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The Impact of Human Activities on Soil Organisms of the Maritime Antarctic and the Introduction of Non-Native Species in Antarctica

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The Impact of Human Activities on Soil Organisms of the Maritime Antarctic and the Introduction of Non-Native Species in Antarctica

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Abstract

The present study aimed at determining (1) the influence of human activities on Antarctic soil-organism communities as well as (2) the potential introduction of non-native species into Antarctic habitats. In the Antarctic summers of the years 2009/2010 and 2011/2011, soil organisms (plants and the soil fauna of the groups Nematoda, Tardigrada, Collembola, Actinedida, Oribatida und Gamasina) were collected from anthropogenically influenced and non-influenced areas of a total of 13 localities and compared. Introduced non-native plant species could not be determined. Eight species of Collembola and Actinedida recorded especially from Deception Island and Neko Harbour could be determined to be potentially non-native. Although the results were conflicted by high data variability, a significant human influence on the soil fauna could be determined. Human influence mostly led to reduced densities. The human impact was strongest in areas of sporadic vegetational cover. The reactions of individual species were different, which indicates changes in community structure and thus in the ecological function of the soil fauna. Specific recommendations for an improved protection of Antarctic ecosystems from human influence are derived from the results. Particularly called for is an intensification of biosecurity measures against the introduction of non-native species as well as an expansion of specific microhabitats in “no-go” areas within Visitor Site Guides. A limitation of sites allowedly visited by tourists is necessary, as is the installation of an international, long-term soil-biological monitoring program.

Kurzbeschreibung

Die vorliegende Studie hatte zum Ziel, (1) die Auswirkung menschlicher Aktivitäten auf antarktische Bodenorganismengemeinschaften sowie (2) die potentielle Einschleppung von in der Antarktis nicht-einheimischen Arten zu ermitteln. Im antarktischen Sommer der Jahre 2009/2010 und 2010/2011 wurden Bodenorganismen (Pflanzen und Bodentiere der Gruppen Nematoda, Tardigrada, Collembola, Actinedida, Oribatida und Gamasina) aus insgesamt 13 Gebieten in von Menschen beeinflussten und unbeeinflussten Arealen erfasst und verglichen. Eingeschleppte, nicht-einheimische Pflanzenarten konnten nicht festgestellt werden. Bei den Collembola und Actinedida wurden acht Arten hauptsächlich auf Deception Island und Neko Harbour als potentiell nicht-einheimisch identifiziert. Obwohl die Ergebnisse durch hohe Datenvariabilität überlagert waren, konnten signifikante Auswirkungen des Menschen auf die Bodenfauna nachgewiesen werden. Die Beeinflussung durch Menschen führte meist zu verringerten Individuendichten. Der Einfluss von Menschen war bei mittlerer Vegetationsbedeckung am stärksten. Die Reaktion von einzelnen Arten war unterschiedlich, was auf Veränderungen in den Gemeinschaftsstrukturen und somit in der ökologischen Funktion der Bodenfauna hinweist. Aus den erzielten Ergebnissen werden konkrete Empfehlungen für einen verbesserten Schutz antarktischer Ökosysteme vor menschlicher Beeinflussung abgeleitet. Hierzu gehört eine Intensivierung von Präventivmaßnahmen gegen eine Einschleppung nicht-einheimischer Bodenorganismen sowie eine Ausweitung der für Besucher geschlossenen Bereiche um Areale spezieller Mikrohabitate. Eine Einschränkung der Gebiete, die Touristen besuchen dürfen, ist erforderlich, ebenso wie die Etablierung eines internationalen, langfristigen bodenbiologischen Monitoringprogramms.

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Abbreviations

ANCOVA	Analysis of Covariance
ANOVA	Analysis of Variance
ASPA	Antarctic Specially Protected Area
ATCM	Antarctic Treaty Consultative Meeting
ATCP	Antarctic Treaty Consultative Party
ATME	Antarctic Treaty Meetings of Experts
BAS	British Antarctic Survey
CAP	Canonical analysis of principal coordinates
CEE	Comprehensive Environmental Evaluation
COMNAP	Council of Managers of National Antarctic Programs
EBA	Evolution and Biodiversity in the Antarctic (biological datenbank of the Australian Antarctic Data Centre)
EIA	Environmental Impact Assessment
IAATO	International Association of Antarctica Tour Operators
IEE	Initial Environmental Evaluation
MANOVA	Multivariate Variance Analysis
NGO	Non-governmental organisation
NMDS	Non-metric multidimensional scaling (statistical ordination procedure)
PERMANOVA	Permutational MANOVA
PERMDISP	Test of the homogeneity of dispersion in PERMANOVAs
SCAR	Scientific Committee on Antarctic Researd
SMNG	Senckenberg Museum of Natural History Görlitz
TAF	Tiethanolamin-Formalin solution
UBA	Federal Enviroment Agency (Umweltbundesamt)

1. Introduction

Antarctic ecosystems are under pressure and influence from growing anthropogenic sources originating from research and station personal, tourists and similar recreational visitors. Modern touristic travel in Antarctica began towards the end of the 1960s with the deployment of the ship MS Lindblad Explorer, which was built expressly for this purpose and could carry approximately 100 passengers (www.expeditions.com). The number of tourists has subsequently increased from a few hundred per year to many tens of thousands (Fig. 1). Since 1989/1990 the number of visitors to Antarctica has increased exponentially, whereby the numbers have doubled every five years between 1997 and 2007 (Roura et al. 2008, Lynch et al. 2009).

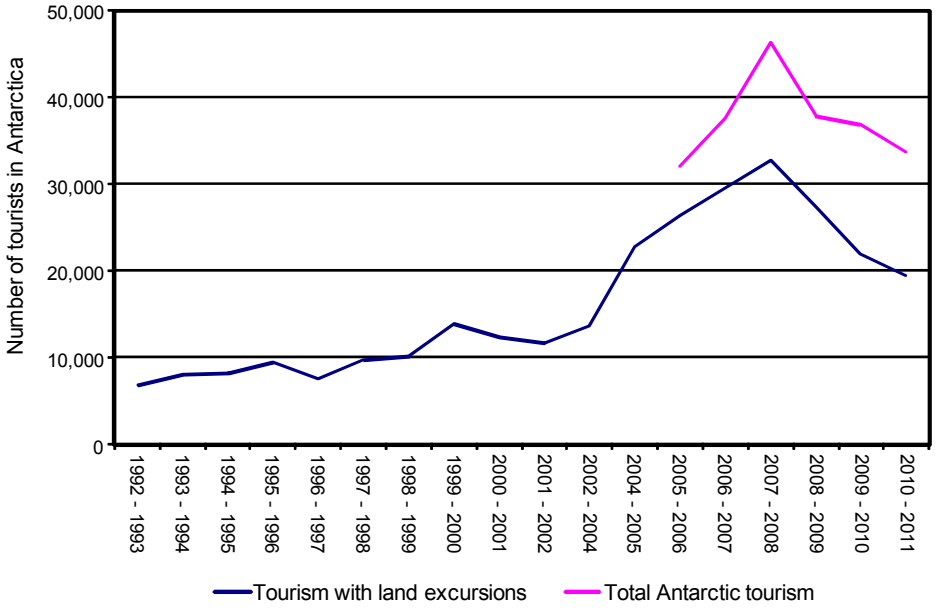


Fig. 1: Development of the number of tourists (excluding ship crew) in Antarctica south of 60° S during the last 20 years. source: IAATO 2011, IAATO Fact Sheets 2009-2010, 2010-2011

Antarctic tourism is not equally distributed throughout all touristic destinations. The increase in the number of visitors has been concentrated in only a few regions (e.g., the South Shetland Islands and the north-western Antarctic Peninsula) and specific sites, e.g., Whalers Bay (Deception Island), Half Moon Island (which has experienced an increase in tourism of over 600% between 1999 and 2005) or Neko Harbour (Naveen et al. 2001; Argentina 2006). While in the austral summer of 1989/1990 less than 2000 tourists visited all Antarctic sites together, 10 years later approximately 20 specific localities were each visited by at least 2000 tourists and eight sites were visited by over 10,000 tourists. Port Lockroy (Goudier Island) and Half Moon Island received more than 15,000 visitors in the season 1999/2000 (Lynch et al. 2009). In the year 2010 these numbers had again substantially increased (Table 1).

The number of tour operators and recreational ships has also increased dramatically in the past decades (Roura et al. 2008). Now more than 40 different operators from almost 15 different countries with approximately 50 cruise ships and yachts offer cruises with the possibility of land excursions, which concentrate on the Antarctic Peninsula (IAATO

2011). Antarctica has thus become a popular goal for tourists and tours in the region are actively marketed. Not only have the number of travellers and ships to Antarctica thereby intensified, but also the time period of touristic activities within the Antarctic summer (Lynch et al. 2009; Fig. 2). Most likely due to the recent global economic difficulties, the number of tourists to Antarctica have decreased somewhat since the 2007/2008 season and is slowly stabilizing around 20,000 and 34,000, respectively, in the season 2010/2011 (Fig. 1). It is expected that the Antarctic ban on ships operated with heavy oil, which came into effect in 2011, will at least temporarily dampen a renewed increase in the number of Antarctic visitors.

Table 1: Number of visitors in selected Antarctic locations in the season 2010/2011 (the 20 most frequently visited sites plus a number of potential study sites in the present study). Source: IAATO 2011

Season 2010-2011	Total
Cuverville Island	29,690
Neko Harbor	25,264
Lemaire Channel	22,504
Whalers Bay	19,477
Elephant Island	19,326
Goudier Island	19,000
Gerlache Strait	18,007
Neumayer Channel	17,312
Paradise Bay	16,273
Palmer Station	15,789
Half Moon Island	15,509
Antarctic Sound	14,895
Petermann Island	12,982
Deception Island	12,264
Pléneau Island	11,317
Jougla Point	11,303
Bismarck Strait	10,715
Almirante Brown	10,200
Aitcho - Barrientos Island	9,591
Vernadsky Station	9,517
Paulet Island	5,301
Telefon Bay	5,101
Melchior Islands	5,066
Detaille Island	4,024
Admiralty Sound	3,555
Devil Island	3,224
Hannah Point	2,719
Point Wild	2,607
Lemaire Island	1,899
Yalour Islands	1,889
Arctowski Station	960
Ardley Island	524
Laurie Island	425
South Orkney Islands	60

The number of persons present in Antarctica for reasons of research is far less than the number of tourists. Currently, 80 research stations are operated by 29 nations in Antarctica, almost half of which are active year round (www.scar.org/information; www.comnap.aq). Many stations have existed for several decades, some continually active since the end of the 1950s, so that the long-term burden on the environment due to these research stations can be considerable. Such environmental impacts are mostly caused by physical changes in the landscape, but also by extensive pollution due to, e.g., accidents, handling of fuels and chemicals or waste products (Kennicutt et al. 2010, Tin et al. 2009). Such environmental contamination in the vicinity of research station has been studied for over 30 years and restoration measures have been carried out time and again (e.g., Peter et al 2008, 2013). In the context of the present study, on the other hand, the focus of interest is in primarily changes in soil substrates and soil organisms due to human trampling. In Antarctica approximately 4,300 people live and work during the summer, reducing to a little over 1,000 in winter (Council of Managers of National Antarctic Programmes: www.comnap.aq). The environmental burden caused by these research personnel can therefore be regarded as being less than that of tourists. However, trampling affects in the vicinity of research stations is considerably more concentrated. The cumulative effects connected with research-station personnel has already been the subject of specific studies (e.g., Campbell et al. 1998, Ayers 2008, Chwedorzewska & Korczak 2010). On the other hand, the potential environmental impacts by tourists are spatially much more extensive and thus potentially concern much larger areas especially on and around the Antarctic Peninsula.

Concern regarding the potential impact of tourists on the Antarctic environment was expressed as early as 1966 at the ATCM (Antarctic Treaty Consultative Meeting) (Murray & Jabour 2004). Nevertheless, towards the end of the 1980s, the potential environmental impact of tourists was considered to be minor due to the relatively low number of visitors at that time (Erize 1987, IAATO 2001). Legal protection to the Antarctic environment was first provided by the „Protocol on Environmental Protection to the Antarctic Treaty“ from 1991 (which came into force in 1998). Paragraph 8 of this protocol required an assessment of the effects of all activities on the Antarctic environment, in which tourism was expressly mentioned. However, this assessment was mainly expected to take place by self-regulation of the tourist industry (Maher 2005). Such self-regulation was undertaken almost exclusively by the International Association of Antarctica Tour Operators (IAATO), a US-American non-governmental organization (NGO) founded in 1991 by seven Antarctic tour operators. The IAATO presently has over 75 members from 14 different nations, which mostly offer ship cruises but also plane flights and land-based travel. The IAATO represents almost 90% of all Antarctic tour providers (IAATO 2010). The remaining 10% constitute operators of private and commercial yachts or similar smaller ships, with approximately only half of these authorized by the Antarctic Treaty States (IAATO 2010). The IAATO members commit themselves to compliance with the (environmental) guidelines of the ATCMs as defined in Resolution 4 (2007) of the Antarctic Treaty, such as limitation to the maximum number of tourists (100) that can simultaneously undertake land excursions or the ban on land excursion for ships carrying more than 500 passengers. The self-regulation of the tourism industry concerning the environmental impacts of tourism activities has been assessed as being successful (ATME 2004, Maher 2005) and is considered to be an essential component of the regulation and control of Antarctic tourism. The IAATO has in many aspects preceded the ATCPs (Antarctic Treaty Consultative Party), e.g., concerning tourism management, data collection as well as the development of behavioural guidelines for land excursions (Maher 2005). For example, guidelines of the ATCMs were partly developed by the IAATO itself, IAATO members abided by these guidelines well before they became obligatory for all Antarctic visitors, and the specific IAATO guidelines also partly exceed in content internationally compulsory guidelines.

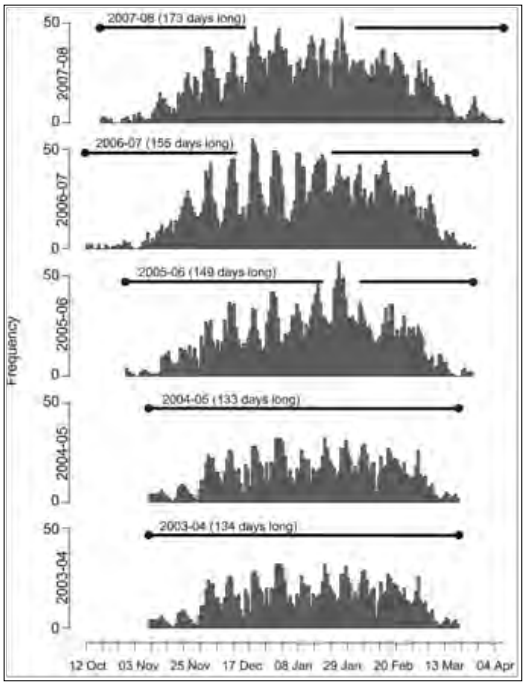


Fig. 2: Increase of the time period per year of touristic activities in the maritime Antarctic. From: Lynch et al. 2009

As part of the environmental protection efforts in Antarctica, the first formalised visitor guidelines were established in 1990 by founding members of the IAATO. At the XVIII. ATCM in 1994, the IAATO guidelines (which had already been modified) were adopted as the basis of Recommendation XVIII-1 (“Guidance for Visitors to the Antarctic” and “Guidance for those Conducting Tourism and Non-Governmental Activities in Antarctica”). These guidelines were intended to limit damage to flora and fauna caused by tourism. Binding, site-specific guidelines for tourists („Visitor Site Guides“) were first adopted at the ATCMs 2005 and 2006 (ATCM 2005,

2006). These guidelines existed for only 16 localities, but were continually supplemented throughout the following years. Presently (in 2012), 35 such guidelines exist for the most frequently visited localities. However, more than 200 sites are regularly visited by tourists (http://iaato.org/de/tourism-statistics). These guidelines primarily contain behavioural rules, specify local visitation zones and serve as protection of the environment; predominantly the protection of breeding colonies (penguins, seals and sea birds) as well as extended patches of vegetation. Other environmental aspects, such as soils or even soil organisms, are not mentioned at all.

These visitor guidelines primarily aimed toward preventing environmental damage and are based on Article 3 of the Environmental Protocol, which states that human activities are to be planned and carried out such that negative impacts to the environment remain limited. According to Article 8 of the Environmental Protocol, an Environmental Impact Assessment (EIA) is required for all human activities in Antarctica, whereby different levels are designated (s. Annex I). In the so-called preparation phase, a preliminary evaluation of expected environmental effects is to be carried out. If only slight or temporary disturbances are thereby predicted, then an Initial Environmental Evaluation (IEE) takes place. If more serious impacts are predicted, then a Comprehensive Environmental Evaluation (CEE) is necessary. However, impacts predicted in the preparation phases are rarely followed up or verified by the Antarctic Treaty Consultative Parties. EIAs are usually only adopted by singular activities and the EIA-Instruments are rarely activated (Tin et al. 2009). Up to now, EIAs of the effects of Antarctic tourism mostly concerned preparatory phases or at the most IEEs and were sometimes superficial and inadequately implemented. Furthermore, they rarely considered the risk of *cumulative* impacts of tourism (Kriwoken & Rootes 2000, Hemmings & Roura 2003, Roura et al. 2008), although this is specifically mentioned in Article 3 of the Environmental Protocol. An extraordinary meeting of the Antarctic Treaty States in 2004 on the effects of Antarctic tourism acknowledged the lack of binding standards as well as the inadequate control and execution of EIAs (ATME 2004). CEEs of touristic activities have to date not been carried out at all (Tin et al. 2009). Correspondingly, no official monitoring of the impacts of tourism on the Antarctic environment exists. Any knowledge of such impacts is based exclusively on single scientific studies. Specific studies focussing on the impact of touristic activities on Antarctic soils and soil organisms are thus lacking and badly needed.

The majority of scientific investigations of anthropogenic influences on the Antarctic environment deal with marine ecosystems (Knox 2006, Kennicutt et al. 2011) or with the consequences of chemical contamination caused by oil leaked during ship accidents, by garbage dumps or by wastewater (Poland et al. 2003, Bargagli 2005, Peter et al. 2008, Tin et al. 2009, Kennicutt et al. 2011). Although such contamination is widely distributed throughout Antarctica, it causes are generally connected with research stations and therefore usually lie in the direct vicinity of such stations (Tin et al. 2009). Objects of terrestrial research as well as the target of most touristic excursions are the widely distributed breeding and moulting sites of vertebrate animals such as penguins, seals and seabirds. Therefore, previous studies have mainly investigated the reaction of Antarctic invertebrates (penguins, seabirds and (partly) seals) to the presence of humans, both regarding station personnel as well as tourists. Anthropogenically caused reactions of these animals can, however, be species-, site- and activity specific (Holmes et al. 2006), so that the results of these studies are partly contradictory (de Villiers 2008). Although human activity apparently causes little disturbance of, e.g., Adélie

(*Pygoscelis adeliae*) or Gentoo penguins (*Pygoscelis papua*) (Cobley & Shears 1999, Carlini et al. 2007, Trathan et al. 2008), indications do exist for anthropogenically caused population declines of Southern Giant Petrel (*Macronectes giganteus*) and Snow Petrel (*Pagodroma nivea*) (Chen & Blume 1997, Micol & Jouventin 2001, Peter et al. 2008, Tin et al. 2009). However, previous data did not show clearly an anthropogenic impact on population densities, so do not indicate that tourism has a significant impact on Antarctic wildlife, but rather imply that potential impacts are often not recognizable due to the influence of other highly variable environmental factors (Fraser & Patterson-Fraser 2009). Other parameters sometimes show more distinct changes caused by human activities, such as the behaviour or physiology of the animals (Pfeiffer & Peter 2003, de Villiers 2008, Peter et al. 2008). Potential impacts of multiple stressors and cumulative effects on the populations or biology of the species studied can also be aggravated by ongoing tourism (Micol & Jouventin 2001, Holmes et al. 2006). Anthropogenically caused impacts on the Antarctic environment are therefore sometimes subtle, indirect or first noticeable after longer periods and must always be evaluated in relation to other influencing environmental factors.

The preferred Antarctic breeding and moulting sites of penguins, seals and sea birds are generally ice-free areas. However, only about 1% of the entire Antarctic continent is free of ice and 60-80% (depending on literature source) of this minimal ice-free area is found on the Antarctic Peninsula (Fox et al. 1994, Beyer & Bölter 2002). Only in such ice-free areas can soils and, accordingly, soil organisms be found. Since the concentrations of touristic excursions target exactly such areas, these sites are particularly susceptible to disturbance and negative impacts by human activities. Important processes of soil genesis (e.g., weathering of bedrock, development of mineral soil, input of organic matter, bioturbation or soil horizon development), which consist of diverse biological and abiotic processes in the temperate zones (Bardgett 2005) are, in Antarctica, reduced to only a few, often purely abiotic processes or do not exist at all (Bayer & Bölter 2002). Accordingly, Antarctic soils are considered to be very sensitive to anthropogenic disturbance (Campbell et al. 1998, Beyer & Bölter 2002). Despite these facts, only few studies on the impact of human activities on Antarctic soils and soil organisms have been carried out in the past. In addition to chemical contamination (e.g., Chen & Blume 1997), some studies have explored the effects of direct physical changes by vehicles or individuals, for instance rut formation, erosion or soil compaction (Hofman & Jatko 2000, IAATO 2001, Campbell et al. 2008, Tejedo et al. 2009, Naveen & Kynch 2011). Terrestrial Antarctic habitats are especially sensitive to such disturbances, which can be compounded by continual human activities (cumulative effects; IAATO 2001, Tejedo et al. 2009) and by the very limited natural recovery and regeneration rates in Antarctic soils (Campbell et al. 2008, Tejedo et al. 2009).

The Antarctic terrestrial soil flora and fauna is very species poor, whereby many higher taxonomical groups commonly found in more moderate climatic zones (e.g., Lumbricidae, Diplopoda) are completely missing (Convey 2005, 2011). The biodiversity and complexity of terrestrial species communities generally become more reduced with increasing latitude and climatic harshness, although the small-scale variability is very high (Chown & Convey 2007, Peat et al. 2007). In the Antarctic flora, botanical species communities are generally comprised of cryptogams (mosses, liverworts and lichens) with only two higher (valcular) plants species occurring naturally along the Antarctic Peninsula (Frenot et al. 2005, Peat et al. 2007). The few studies of anthropogenic influences on the vegetation mostly attest to direct damage, e.g.,

trampling of mosses or the grass *Deschampsia antarctica*, or to indirect damages caused by an impact on growth parameters or the underlying soil material. Such damage often occurs most strongly in regions of the Antarctic Peninsula (Hofman & Jatko 2000, IAATO 2001, Bargagli 2005, Peter et al. 2008, Naveen & Kynch 2011). Mosses and especially lichens can be damaged by the emission of pollutants, even if the emission source (usually research stations or ships) are over 100 km distant (Chen & Blume 1997). Such negative impacts can be considerably more damaging than in temperate regions, because even relatively minor human influence can occur to a degree that is beyond the natural regeneration ability of the vegetation (Tin et al. 2009).

Ice-free areas of Antarctica are particularly important for the terrestrial invertebrate faunal communities, because such species can only find Antarctic ecosystems habitats under or between stones, or in soil substrates. Besides tropical forests and coral rifts, soils represent one of the largest reservoirs of animal biodiversity on earth (André et al. 1994, Giller 1996). The terrestrial invertebrate species occurring in Antarctica are generally relatively well known due to the many, albeit often purely descriptive studies of the last 120 years (e.g., Michael 1895, Trägårdh 1908, Dalenius & Wilson 1958, Womersley & Strandtmann 1963, Wise 1964, 1971, Wallwork 1965, 1973, Hunter 1967, Greenslade & Wise 1984, Somme 1986, Usher & Booth 1986, Dastych 1989, Potapov 1991, Higashi & Sugawara 1992, Greenslade 1995, Block & Stary 1996, Andrássy 1998, McInnes & Pugh 1998, Convey et al. 2000a, Sanyal & Gupta 2005). Many major taxonomical groups that are usually found in soils are totally missing in Antarctica and the endemic terrestrial invertebrate fauna merely consists of Diptera (flies, albeit only two species), Acari (mites), Collembola (springtails), Nematoda (roundworms), Rotifera (rotifers), Tardigrada (water bears) and Protozoa (single-celled animals) (Block 1984, Hogg & Stevens 2002, Convey 2005). Compared with other ecosystems, e.g. in the tropics or the temperate zones, the terrestrial Antarctic fauna is very species- and structurally poor throughout all occurring animal groups (Ryan et al. 1989, Sohlenius et al. 1995, Block & Stary 1996, Convey & Lewis Smith 1997, Freckman & Virginia 1997, Convey et al. 2000b, Convey 2005, 2011). The faunal communities, especially of Continental Antarctica, belong to the simplest on earth (Freckman & Virginia 1997, Convey et al. 2000b, Hogg et al. 2006), whereby even the otherwise ubiquitous Nematoda can be totally missing (Convey & McInnes 2005). Nonetheless, the individual numbers of species occurring in Antarctic ecosystems can at times be very high (Ohyama & Hiruta 1995, Sohlenius et al. 1995, Caruso & Bargagli 2007, Sanyal & Hazra 2008, Schulte et al. 2008, Sohlenius & Bostrom 2008), so that these very simple biotic communities can be composed of very few species in very large populations. Furthermore, the functional diversity of Antarctic terrestrial habitats is very limited. Most Antarctic soil invertebrates are most likely microbivorous or detritivorous, while true herbivores and predators only play a very minor role (Convey et al. 2000b, Hogg et al. 2006, Tin et al. 2009).

Although many scientific basic studies have been carried out on the Antarctic soil fauna, hardly any studies of the impact of human activities on Antarctic terrestrial invertebrate communities exist. The few existing studies have at times shown negative effects even after only minor anthropogenic influence (Ayers et al. 2008, Tejedo et al. 2009). However, these studies only investigated the impact on total densities of single species or only major groups. An impact on other biodiversity parameters remains unknown.

In contrast, the potential or actual introduction of non-native species into Antarctic habitats has been more intensely studied. A clear danger of an anthropogenic introduction of species into

Antarctica habitats, especially by tourists, has often been emphasized in the past (Bergstrom & Chown 1999, Hofman & Jatko 2000, de Poorter et al. 2006). On the one hand, Antarctic species are particularly adapted to the extreme climatic conditions, so only species that survive the stresses of severe cold and aridity can occur (Somme et al. 1993, Block & Harrisson 1995, Wharton 2003). Species immigrating from lower latitudes usually cannot survive these conditions (Convey et al. 2000b, Sinclair 2002). On the other hand, the low biodiversity and simplicity of Antarctic terrestrial biotic communities potentially render them very susceptible to colonization by immigrating species; firstly, because the indigenous species are most likely very competition-poor and have little protection against predators (Frenot et al. 2005, Convey 2006) and, secondly, previously unoccupied ecological niches (i.e., ecosystem functions or trophic levels that had not previously existed) can be filled by newly introduced species (Tin et al. 2009, Convey 2011).

In the past there have been a few intentional anthropogenic introductions of non-native plant or animal species (i.e., during botanical transplantation studies), but for the most part introductions have been unintentional, via freight, vehicles, food or clothing of station personnel and tourists (Downie et al. 2000, Frenot et al. 2005, Tin et al. 2009, Hughes et al. 2011). To date, five plant species, three of which are grasses (*Poa annua*, *P. pratensis*, *P. trivialis*), are known to have been introduced into Antarctica and to have established permanent populations in the vicinity of research stations (Frenot et al. 2005, Tin et al 2009). Among the introduced animal species, two were most likely to have been imported during plant transplantation experiments on Signy Island: the Enchytraeid *Christensenidrilus blocki* and the Chironomid *Eretmoptera murphyi* (Convey & Block 1996, Dózsa-Farkas & Convey 1997, Frenot et al. 2005. Tin et al. 2009). Both are also known from the neighbouring Subantarctic South Georgia. Otherwise, only collembolan species have been identified as being introduced, mostly in the Maritime Antarctica (South Shetland Islands, especially Deception Island, and Marguerite Bay): *Hypogastrura viatica, Folsomia candida* and an as yet unidentified *Protaphorura* species (Greenslade & Wise 1984, Greenslade 1995).

A further danger to Antarctic soil organisms exists in the impact of anthropogenic transfer of species between different localities of the continent. Antarctica does not consist of a single unified biogeographical region. The so-called „Gressitt Line“ (Chown & Convey 2007, Tin et al. 2009, Convey 2011), a biogeographical border crossing the southern Antarctic Peninsula, marks an almost absolute break in species inventory and exists for a few taxonomically important major groups (e.g., Acari, Collembola, Nematoda). A strong regionalized occurrence of various species exists in all areas of the Antarctic continent and the Antarctic Peninsula (Chown & Convey 2007, Caruso & Bargagli 2007, Convey 2011). These significantly fragmented and isolated species communities offer ideal conditions for local differentiation of single species and communities (Chown & Convey 2007, Tin et al. 2009). In specific areas, even taxonomical important major groups are missing, i.e. Collembola on Nunataks of Charcot Island (Convey et al. 2000a) or Nematoda in Ellsworth Land (Convey & McInnes 2005). The transfer of other species (especially from these major groups) into areas in which they otherwise do not occur can dramatically change the fragile community structures or even entire soil food webs.

In light of the very large number of tourists visiting ice-free areas of Antarctica, the impact of human, in the most part touristic activities on Antarctic soils and soil organisms can be formidable. Almost all basic scientific studies, as well as the few investigations on the impact of human activities on Antarctic soil organisms, were performed in relatively limited, local areas

and usually concentrated on single taxonomic groups. Investigations of anthropogenic impacts at larger regional scales are extremely rare (Tin et al. 2009). The present study therefore aimed to investigate the potential anthropogenic impacts on the major components of the soil biotic community (with the exception of microorganisms: bacteria and fungi) over a wide region strongly affected by tourism (the Antarctic Peninsula). Localities that are strongly frequented by tourists, and also research-station personnel and scientists, were especially chosen as study sites. As a first project target, potentially introduced non-native soil animal species were to be specifically identified. In light of the danger of an introduction of non-native species, the present study investigated and evaluated the effectiveness of prevention and cautionary measures ("Bootwashing") against unintentional introductions of soil organisms. The present study also investigated the anthropogenic impact on the occurrence and composition of native soil species and invertebrate communities in maritime Antarctic localities. Furthermore, the hypothesis that areas with high tourism pressure could facilitate the transfer of species between microhabitats within a locality, which may be larger than between localities, leading to the homogenisation of the species composition for different microhabitats and thus to a reduction in ß-diversity (= species turnover between micro-habitats) was tested.

2 Materials and Methods

2.1 Fieldwork

2.1.1 Selection of potential study sites

Maritime Antarctica, in which the current project is concentrated, includes (among others) the South Shetland Islands, the South Sandwich Islands, Bouvet Island, the South Orkney Islands, the Palmer Archipelago as well as the western side of the Antarctic Peninsula up to 72° S with its neighbouring smaller and larger islands (Fig. 3).

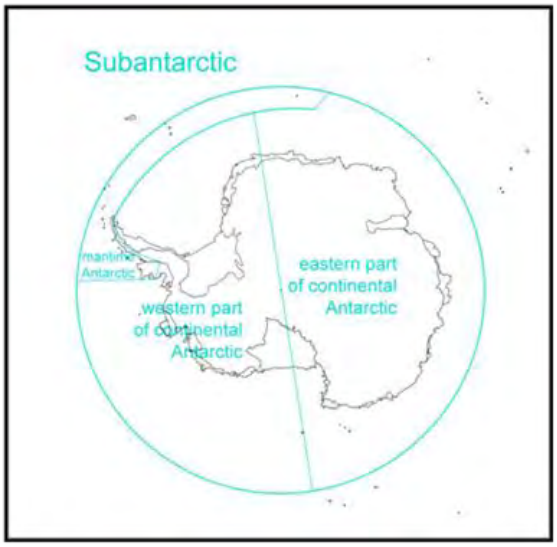


Fig. 3: Division of terrestrial Antarctica into biotic zones in a broad understanding.

Since the key goal of the present study concerned recognizing anthropogenic impacts on soil organisms, the main criterion in the selection of potential study sites was a regular presence of humans, in particular tourists. Especially study sites experiencing high tourist pressure or considerable research activities were necessary. Therefore, predominantly locations along the Antarctic Peninsula visited by touristic cruise ships during land excursions came into consideration for the fieldwork. Furthermore, areas were preferred that were strongly frequented by station and research personnel.

To keep the sampling as cost-efficient as possible, it was planned to carry out the fieldwork during normally scheduled cruise-ship tours and/or during previously planned research trips of other workgroups. Since these are variable from year to year, the exact study sites could not be established in detail at the beginning of the project. The project management (German Federal Environment Agency: UBA) agreed to choosing the specific study sites first after the project had begun. For this, different German tour operators were approached, who agreed to support the fieldwork during land excursions of various Antarctic tours of three cruise ships: the MS Hanseatic, the MS Bremen (both Hapag-Lloyd Kreuzfahrten GmbH Hamburg) as well as the MS Delphin (Hansatours GmbH Hamburg). However, it would have been financially prohibitive and extremely time consuming for project personnel to join all cruises and research trips. It was therefore arranged together with the project management that field sampling be performed by the scientific personnel of the cruise ships and research groups. The following expedition leaders agreed to participate: Dr. Arne Kertelhein (MS Hanseatic), Dr. Klemens Pütz (MS

Delphin), Dr. Hans-Joachim Spitzenberger (MS Bremen). Furthermore, the possibility arose that members of the work group Ornithoecology of Dr. Hans-Ulrich Peter (Institute for Ecology of the Friedrich Schiller University Jena) could sample additional sites on the Fildes Peninsula of King George Island during their annual research trip there between January and March.

Exactly which area was to be sampled and how sampling was to take place depended on the nature and conditions of the potential study sites, in particular the existence of diverse microhabitats (i.e., bare soil substrates, vegetation, melt streams etc.) both influenced by tourists and occurring in a non-influenced state. Based upon the list of probable land-excursion sites of the three cruise ships listed above, information on the climate, geomorphology, soils and ecological conditions of the localities was researched. In spite of obtaining all possible information, the local conditions of the individual excursion sites of these cruise ships was not known in detail at the beginning of this project. For this reason, the expedition leaders of the cruise ships were invited to report on their many years of experience in the known land-excursion localities. During these meetings the expedition leaders were briefed on and received first instructions in soil-zoological sampling methods.

Proceeding from the results of this background research and meetings with the expedition leaders, as well as from the ecological requirements of potential study sites, firstly, tours of the three cruise ships listed above were chosen in which appropriate weather conditions could be expected (i.e., snow cover no longer probable etc.), and, secondly, potential study sites (= localities) among the expected land-excursion sites of these tours were identified. All potential study sites were located on or around the Antarctic Peninsula.

Potential study sites

- King George Island (Arctowski Station)
- Paulet Island
- Devil Island
- Half Moon Island
- Deception Island (Whalers Bay)
- Cuverville Island
- Neko Harbour
- Peterman Island
- Yalour Islands
- Horseshoe Island

Furthermore, various study sites on or around the Fildes Peninsula (King George Island) were taken into consideration. All of these various localities were at first considered to be primarily *potential* study sites, since the actual sampling possibilities in these areas depended upon the current environmental conditions (weather, snow, ice), which determined whether land excursions from the cruise ships or accessibility of study sites by research personnel was possible or not. It was therefore necessary to target various tours of different cruise ships within the Antarctic summer for carrying out the field sampling, in order that as many sampling sites per year as possible could be obtained.

2.1.2 Sampling design within the Antarctic study sites

During studies of soil organisms, it is necessary that the exact number and distribution of soil samples (= data collection) be adapted to the planned statistical data analyses. Therefore, as a first step in determining the sampling design to be used during the fieldwork, the statistical

analysis methods - especially concerning the evaluation of ß-diversity differences - were discussed and coordinated with the European project partners. It was thereby chosen to use a multivariate ANOVA („PERMANOVA“ = permutational MANOVA; s. below: "Statistical Analyses“) with the main factors „anthropogenically influenced“ and „anthropogenically non-influenced“. The sampling design of the fieldwork was adapted to these analysis methods as well as on the expected field conditions.

Based upon the results of the background site research as well as the meetings with the expedition leaders listed above, a detailed sampling design (including exact information on the number and distribution of all soil and vegetation samples) was coordinated and agreed upon with all project partners. A sampling design was collectively defined which (1) took the expected local environmental conditions into consideration, (2) guaranteed the data basis necessary for a robust statistical analysis of the results in regard to the main study questions (especially concerning ß-diversity) and (3) was feasible within the project duration and resources regarding the necessary amount of labour. Due to the expected high ß-diversity (due to habitat fragmentation and isolation of individual species, e.g., Richard et al. 1994, Stevens & Hogg 2002, Caruso & Bargagli 2007), the use of a nested sampling design is recommended for field studies of Antarctic soil organisms (Caruso & Bargagli 2007). The sampling design used in the present study therefore contained the following nested levels of hierarchy (Fig. 4):

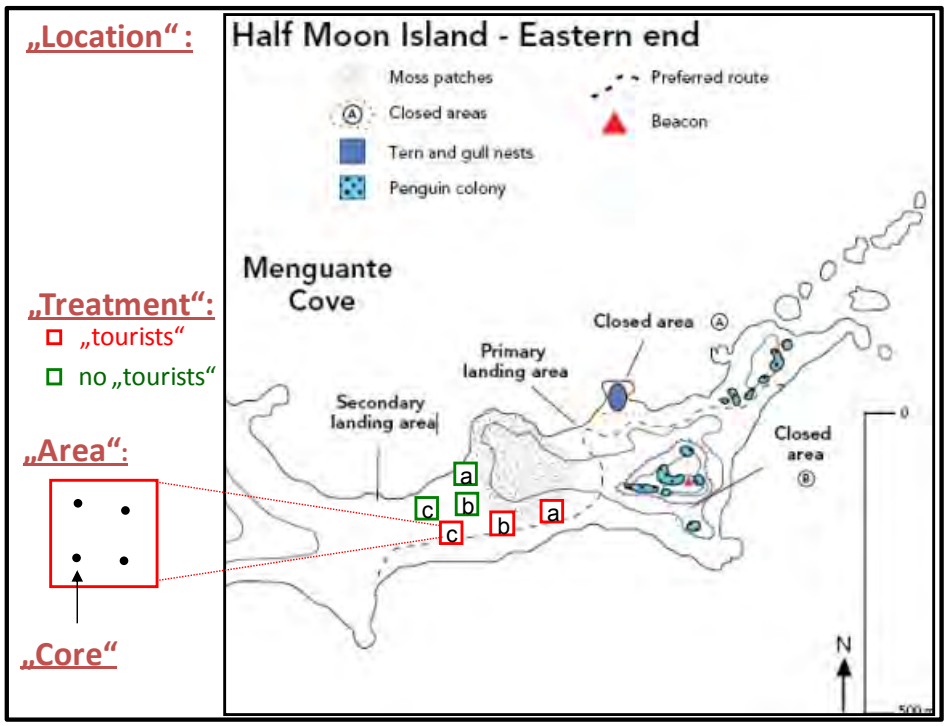


Fig. 4: Levels of hierarchy contained in the sampling design used in the present study, as exemplified by Halfmoon Island (map from the „Halfmoon Island Visitor Site Guide“).

- „Location “ (= in each season maximally nine different Antarctic landing sites of the cruise ships. If more than nine localities were sampled, only those samples from the most promising locations were to be further processed and evaluated during the current project.)
- „Treatment“ (= either anthropogenically influenced or non-influenced areas within a locality.)

- „Area“ (= difference study plots within a treatment; separated into - preferably - three anthropogenically influenced and three non-influenced plots per locality.) The exact choice of the plots was to take place by pairs (one anthropogenically influenced and one non-influenced plot, which should otherwise be similar in regard to soil substrate, vegetation etc.) to ensure comparability of the treatments. Since tourists especially visit Antarctic sites that are strongly frequented by wildlife (seals, penguins and seabirds), careful attention was to be paid for similar wildlife densities in the anthropogenically influenced and non-influenced plot pairs. It was otherwise not possible in the present project to differentiate between an influence of humans and wildlife. This resulted in the disadvantage that only an anthropogenic influence needed could be recognized that was above and beyond that of wildlife.
- „Core“: the specific soil sample within a plot (four per plot, evenly distributed within a square meter). Since soil organisms are per se are very heterogeneously distributed even at very small spatial scales, multiple samples per study plot are necessary in order to ensure that the biotic communities are representatively sampled and not only a random selection of the occurring species are detected. Four samples per square meter was considered to be an absolute minimum, but still possible to process within the project duration. During previous investigations of the Antarctic soil fauna, the undersides of stones are very often washed during sampling (e.g., Goddard 1979, Richard et al. 1994, Convey & Lewis Smith 1997, Convey & Quintana 1997). However, this sampling method is very time intensive and was therefore not possible during the short time periods available during land excursions. The limitation to soil samples was therefore necessitated by the time available for sampling in each locality; however, an important microhabitat for soil fauna was thereby missed. Nonetheless, the uniform sampling method guaranteed the comparability of all study locations. Besides soil sampling, if vegetational differences between plots (areas) were macroscopically recognizable, additional vegetation samples per area were to be taken in order to register the ground vegetation as completely as possible.

2.1.3 Preparatory activities

This sampling design was explained in detail to the expedition leaders and the scientific personnel carrying out the sampling. Furthermore, detailed fieldwork guidelines were developed for those performing the sampling. The guidelines contained a scheme of the necessary spatial distribution of the single samples as well as criteria for choosing sampling plots (= “areas”) and the individual sampling points (Appendix 1, Fig. A1-1). Furthermore, a standardized protocol for the fieldwork was developed together with all project partners, which contained all the information necessary for latter data evaluation and therefore needed to be noted during fieldwork (Appendix 1, Fig. A1-2). This protocol, firstly, guaranteed that sampling was standardized and thus all results were comparable and, secondly, that the sampling was well understood after the fact by the project partners analyzing the data. The protocol therefore contained all information necessary for the subsequent data analysis.

During the meetings with expedition leaders, these were instructed in detail on the methods of soil-biological sampling. Other expedition leaders and scientists were instructed by telephone. The necessary equipment for the fieldwork was provided, consisting of sampling tools (soil corer of 6 cm radius, hand shovel, knife etc.), sample containers for receiving and transporting the soil

and vegetation samples, digital soil thermometer as well as a digital camera for (comparable) documentation the sampling sites.

2.1.4 Actually sampled locations

Sampling was carried out by the scientists and expedition leaders named above. Since (mainly) adverse weather conditions sometimes did not allow land excursions in specific localities during the chosen cruises, not all of the targeted localities could actually be sampled and only very few localities could be sampled in both study years. In the end, a total of 13 localities were sampled, nine in the first study year (2010), seven in the second year (2011) and only three in both study years (Fig. 5).

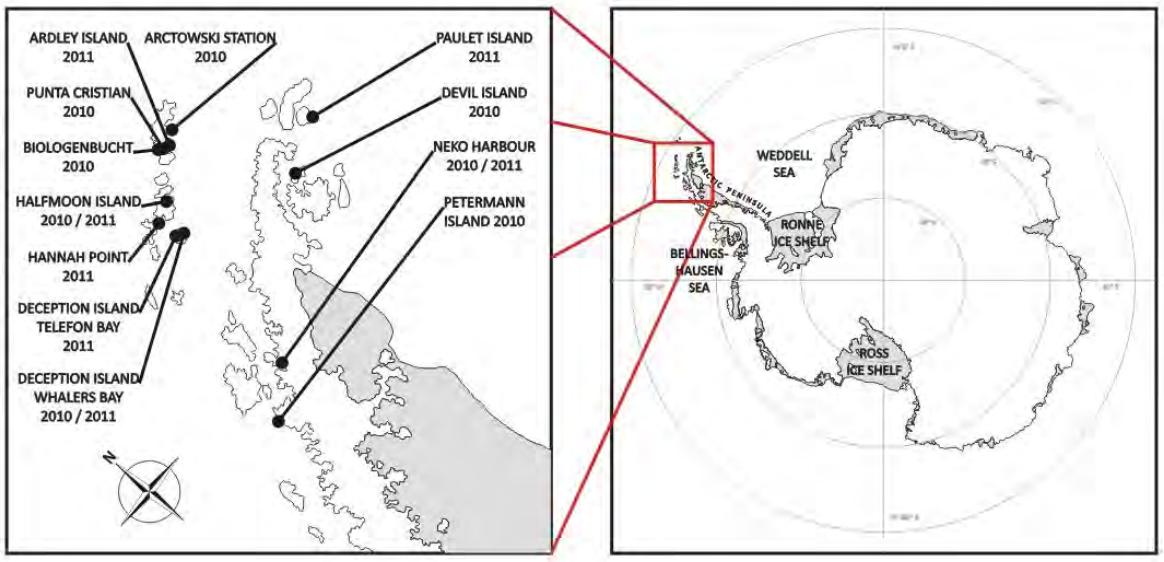


Fig. 5: Maritime Antarctic locations investigated in the current project, with the years in which these locations were sampled.

The northernmost study sites were located in the South Shetland Islands, with many locations on King George Island. The locality „**Arctowski Station**“ (Fig. 6) was situated on the northeast of the island in the vicinity of the Polish research station Arctowski on a peninsula extending into Admirality Bay. The coast there is generally gravelly, whereby the soil substrate contains more clay further inland. A comparatively rich soil vegetation is present, mainly consisting of mosses, lichens as well as the only naturally occurring Antarctic grass species *Deschampsia antarctica*. The influenced areas were frequented both by tourists as well as station personnel. The non-influenced areas were located approximately 5 m distant from the influenced areas within an area of extended vegetation in the Antarctic Specially Protected Area (ASPA) Nr. 128 „Admirality Bay“. The locality was sampled in the study year 2010.

Many study sites were located on the southwest of the island, on or around the approximately 7 km long Fildes Peninsula. The locality „**Biologenbucht**“ is found on the western side of the peninsula south of the Gemel Peaks, approximately 250 m inland of the coast of the bay of the same name (Fig. 7). The sampled areas were located halfway up the southern slope, through which multiple meltwater streams flowed. The soils consisted of a sandy to finely grained substrate interspersed with gravel. Ground vegetation consisted mainly of mosses and was relatively closed. The influenced areas were situated on footpaths occasionally trampled by

station personnel; the non-influenced areas were approximately 50 m distant offside the footpath. The locality was sampled in 2010.

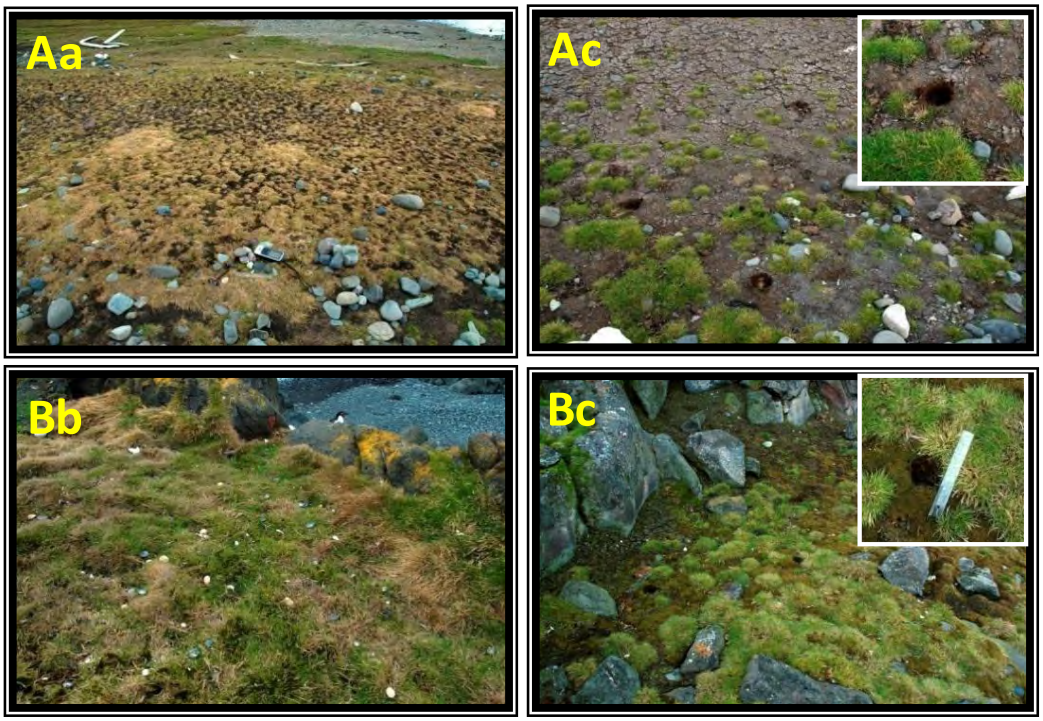


Fig. 6: Sampling plots („areas“) of the locality „Arctowski Station“ on King George Island. Letters represent the individual sampling areas (A = anthropogenically influenced, B = non-influenced). Inserts show a magnification of the specific sampling points.

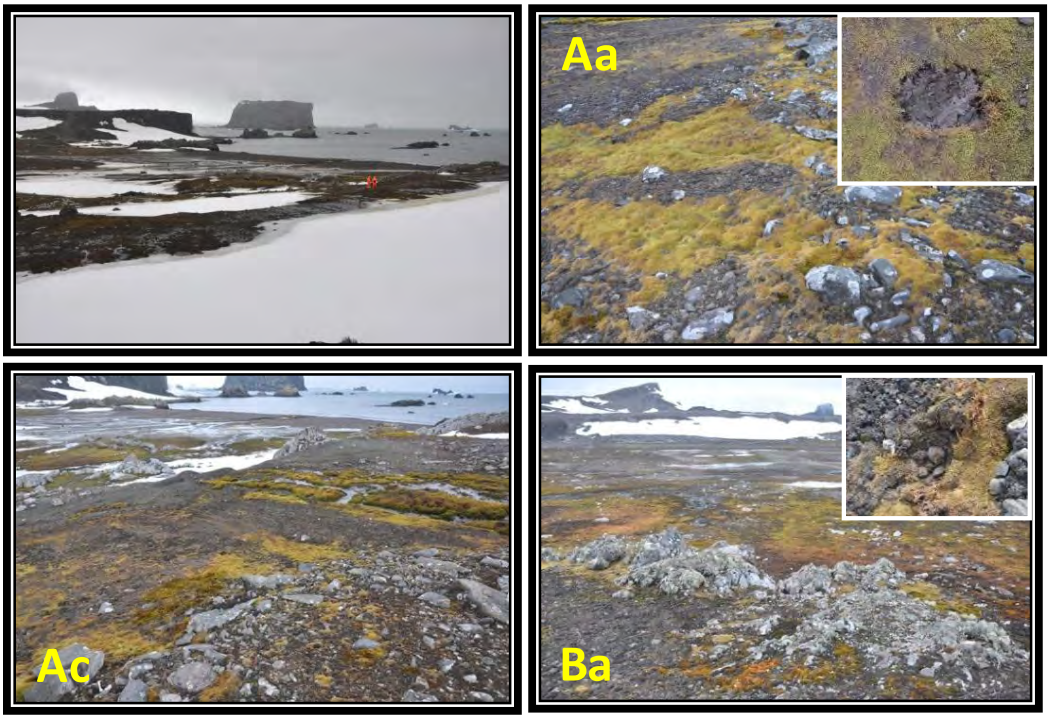


Fig. 7: Sampling plots („areas“) of the locality „Biologenbucht“ on the Fildes Peninsula of King George Island. Upper left = overview of the sampling site. Letters represent the individual sampling areas (A = anthropogenically influenced, B = non-influenced). Inserts show a magnification of the specific sampling points.

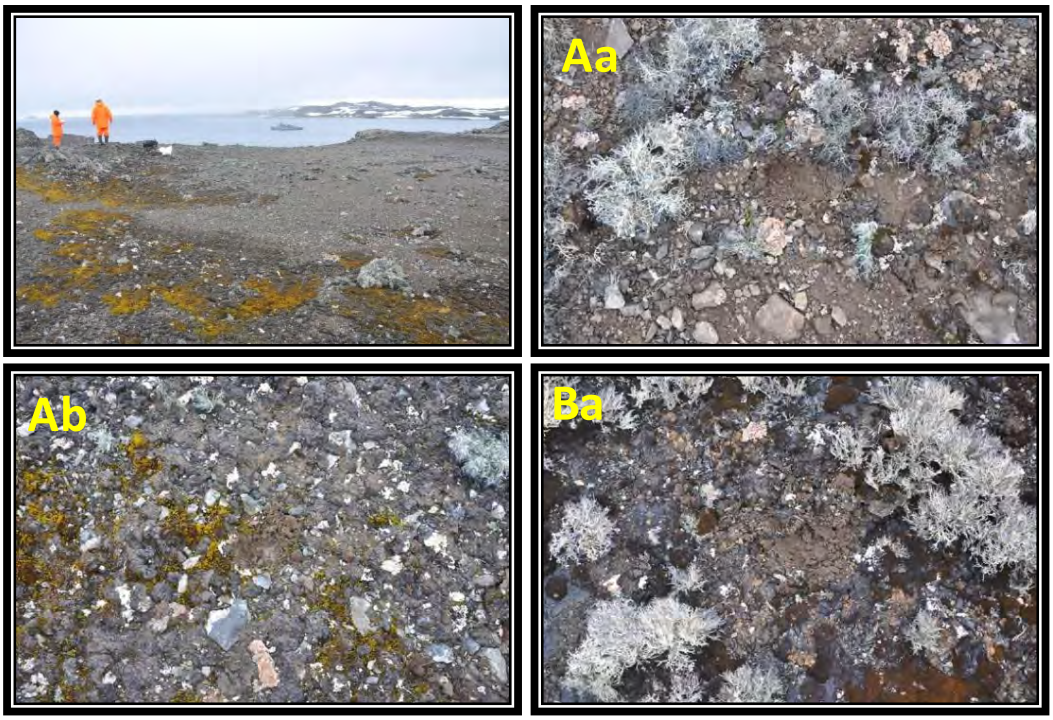


Fig. 8: Sampling plots („areas“) of the locality „Punta Christian I“ on the Fildes Peninsula of King George Island. Figure explanations as in Fig. 7.

Two further study sites were located on the eastern side of the Fildes Peninsula, on the northern coast of Maxwell Bay approximately 1 km from the Russian research station Bellinghausen: „**Punta Christian I & II**“ (Figs 8 & 9). Punta Christian I was located on a cliff above the coast along the foot path leading to Punta Rodriguez. The substrate of this locality was rocky with a very thin sandy soil layer interspersed with gravel. The vegetation consisted of patchily distributed moss cushions as well as lichens on exposed rock. The influenced areas were located along the footpath; the non-influenced areas a few meters offside this path. Punta Christian II was located approximately 250 m from the first site on the lower coastal terrace of the northern Maxwell Bay. The soil substrate was sandy, interspersed with larger stones. The vegetation consisted of an (at times) patchily distributed moss cover. These two localities were also sampled in the year 2010.

The last study area around the Fildes Peninsula was located on „**Ardley Island**“ (Fig. 10), an island on the west side of Maxwell Bay east of the Fildes Peninsula. A major part of the island is environmentally protected (ASPA Nr. 150 „Ardley Island“). The soil substrate consists mostly of crushed rock and gravel, the vegetation of a relatively dense and closed moss carpet. The anthropogenically influenced area was located along an older vehicle track now used by researchers and station personnel as a footpath. The non-influenced area was located approximately 10-20 m distant from this path. Ardley Island was sampled in the year 2011.

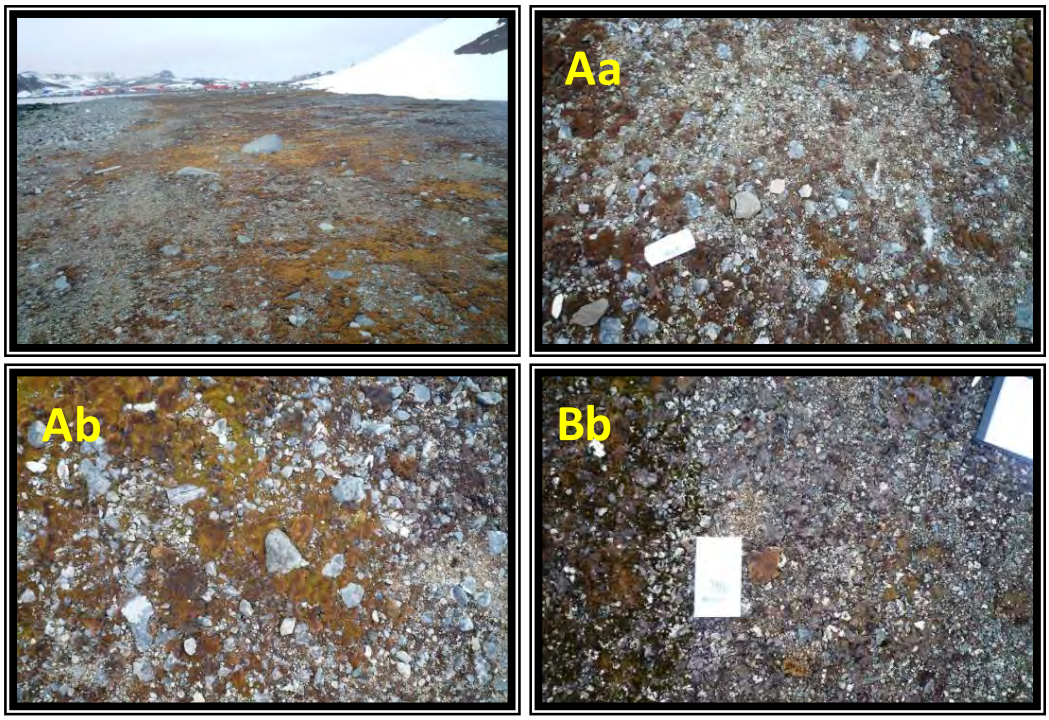


Fig. 9: Sampling plots („areas“) of the locality „Punta Christian II“ on the Fildes Peninsula of King George Island. Figure explanations as in Fig. 7.

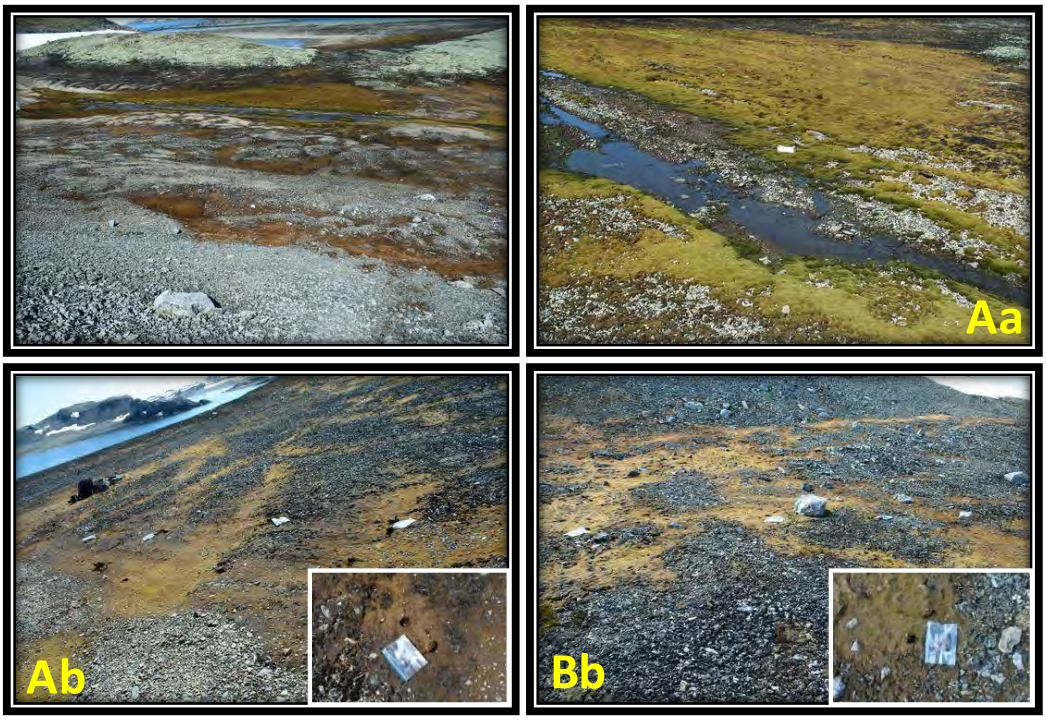


Fig. 10: Sampling plots („areas“) of the locality „Ardley Island“ (selection). Figure explanations as in Fig. 7.

The locality „**Halfmoon Island**“ is a 2 km long crescent-moon-shaped island northeast of Livingston Island. The study site was located in a gravelly area on a peninsula in the southern part of the island. The soils consisted of stones embedded in a clay matrix; the samples were taken from this matrix. The vegetation - if present- consisted of sporadically occurring mosses and lichens as well as single patches of *Deschampsia antarctica*. The influenced area was frequented by tourists. Two areas were closed for tourists due to breeding seabirds and

Chinstrap Penguins (*Pygoscelis antarctica*); an area lightly vegetated with mosses and lichens located on a hill did not belong to the closed area (Naveen & Lynch 2011). The locality could be sampled in 2010 as well as 2011. The influenced areas sampled in 2010 were located along the footpath frequented by tourists; on the one hand on barren soils, on the other at the edge of an area vegetated by mosses (Fig. 11). The non-influenced area was located in 2010 in similar areas approximately 50 m distant from influenced areas. In the year 2011, influenced areas were strongly frequented by tourists and were located at the edge of the penguin colony in „Closed Area B“, the non-influenced area was located at the edge of a penguin colony close to the eastern tip of the peninsula.

Fig. 11: Sampling plots („areas“) of the locality „Halfmoon Island“ (examples). Figure explanations as in Fig. 7.

The locality „**Hannah Point**“ (Fig. 12) was located on a narrow peninsula in the southwest of the neighbouring Livingston Island. The area is very hilly with steep slopes. The underground consists of larger stones and gravel embedded in a clay matrix. Larger rookeries of Chinstrap and Gentoo Penguins (*Pygoscelis antarctica* and *P. papua*) are found on the peninsula. Hannah Point is strongly frequented by tourists due to its geomorphology, large colonies of penguins and seabirds as well as the presence of sea elephants (*Mirounga leonina*) and vegetated areas in an otherwise strongly glaciated region. The actual study sites were located midway up a slope on the rear edge of a Chinstrap Penguin rookery. The influenced areas were found on the edge of footpath strongly frequented by tourists and grown over with the algae *Prasiola crispa*. The non-influenced area was located approximately 5 m further up the slope, where the cover of *Prasiola crispa* merged into a cover of *Deschampsia antarctica*. The locality was sampled in 2010 as well as 2011.

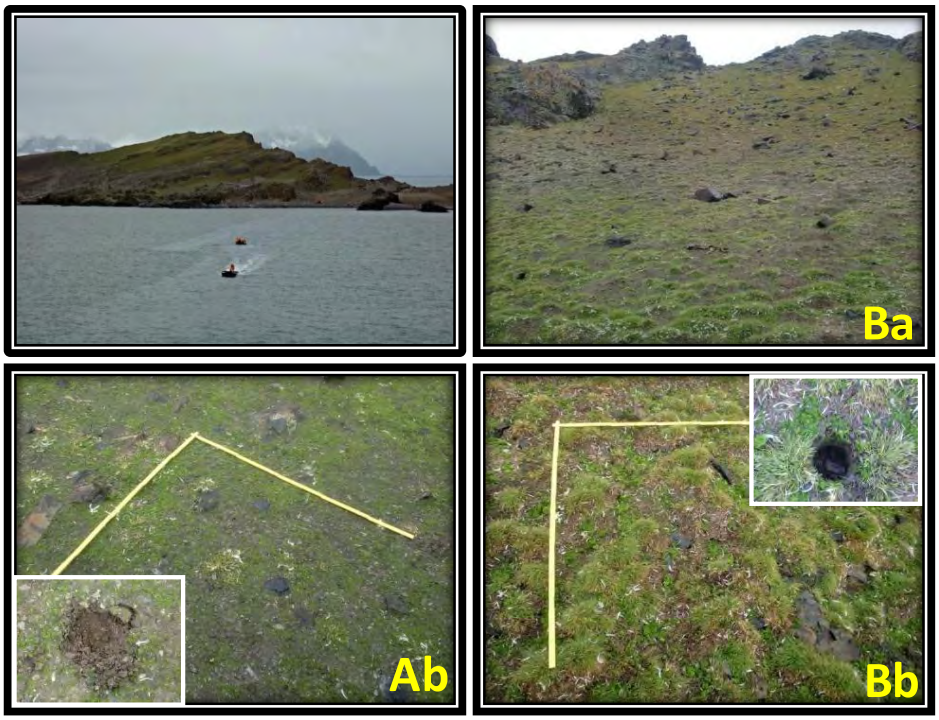


Fig. 12: Sampling plots („areas“) of the locality „Hannah Point“ on Livingston Island (examples). Figure explanations as in Fig. 7.

Two further sites within the South Shetland Islands were located on Deception Island: „Whalers Bay“ and „Telefon Bay“. Deception Island is a circular vulcanic island, the center of which represents the volcano's crater and is filled with seawater. The crater is open and connects with the Antarctic Ocean in the southwest. The last eruption occurred in 1996. The island is still geothermically active and especially the coastal soils towards the crater are very warm and can reach temperatures far above 50° C.

„Whalers Bay“ (Fig. 13) is located on the eastern side of the crater on an extended semicircular coastal terrace consisting of volcanic sand. The area is the site of a whaling station founded at the beginning of the 20th century, the use of which was discontinued at the beginning of the 1930s. Whalers Bay is one of the most strongly touristically visited Antarctic localities (cf. Table 1). This locality is therefore not only strongly frequented by tourists, but also has a long history of human activity. Neither penguin rookeries nor breeding colonies of seabirds or seals are found in this area, most likely due to the strongly warmed soils. Soils of the study area consist of almost purely barren volcanic sands, on which only sporadic patches of mosses, lichens and algae are found. The human influenced area was located in the center of the coastal terrace in the vicinity of the landing site of the cruise ships' zodiacs. The non-influenced area laid 50-100 m further inland in a transitional area between the coastal terrace and the steep slopes rising in the west. The area was sampled in both study years. In 2011 two further areas were sampled, which were located in barren sands on the tourist path to Roland Hill (influenced area) and approx. 50 m south of this path (non-influenced area).

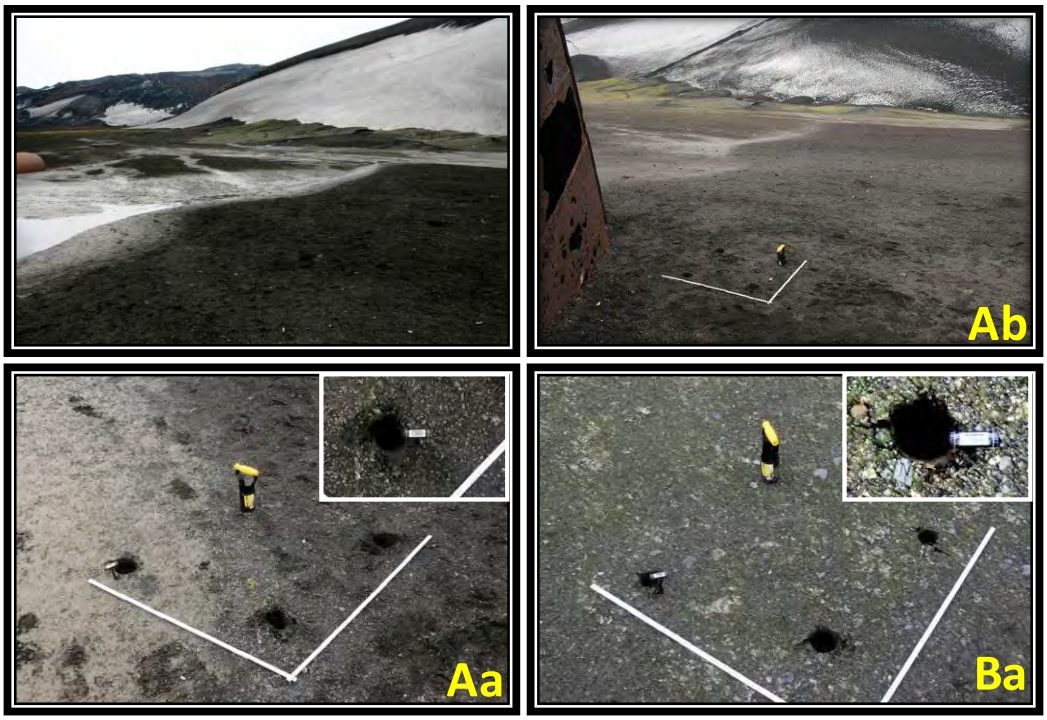


Fig. 13: Sampling plots („areas“) of the locality „Whalers Bay“ on Deception Island (examples). Figure explanations as in Fig. 7.

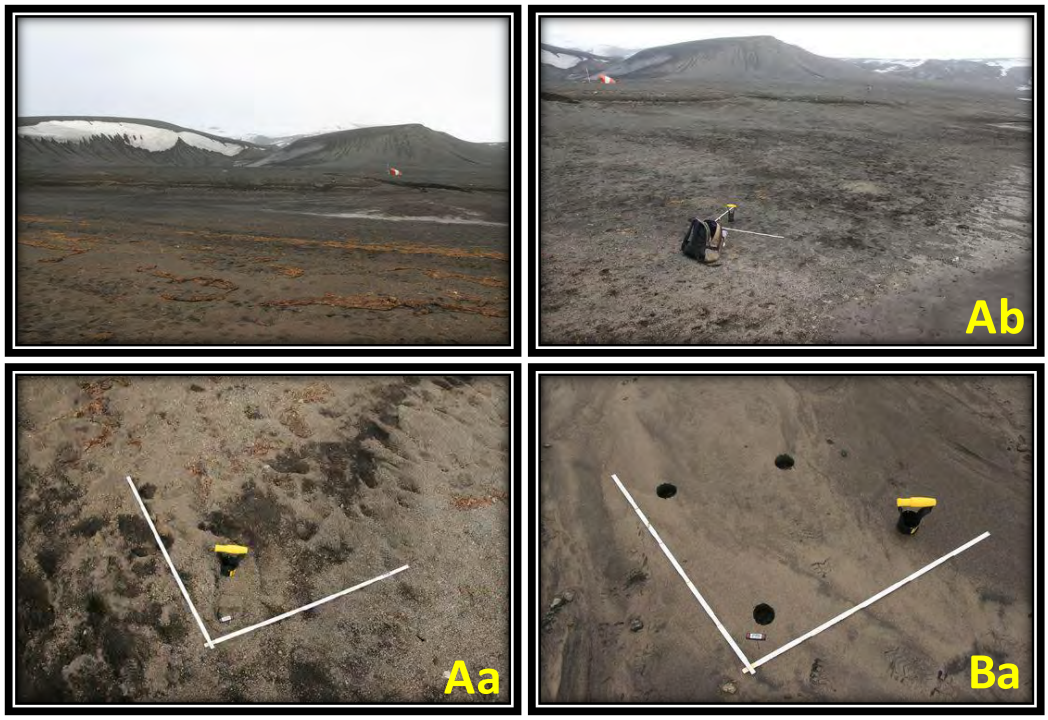


Fig. 14: Sampling plots („areas“) of the locality „Telefon Bay“ on Deception Island (examples). Figure explanations as in Fig. 7.

„Telefon Bay“ (Fig. 14) is located in the coastal area in the north of the inner volcanic crater. The soils of the study site also consist of barren volcanic sands without vegetation and are very warm as well due to the geothermal activity. As in Whalers Bay, many small meltwater streams flow through the area, the exact location of which fluctuate from year to year. This locality is more seldomly visited by tourists and is sporadically entered by researchers studying the succession after the last volcanic eruption (Naveen & Lynch 2011). In this area, no penguin

rookeries or other wildlife colonies are found. The influenced area was located along the footpath near the coast; the non-influenced area approximately 50 m beside the footpath. The locality was only sampled in 2011.

Two study sites were located on islands in the Weddell Sea in the upper northeast of the Antarctic Peninsula near the exit of the Antarctic Sound into the Weddell Sea. The two localities „**Devil Island**“ and „**Paulet Island**“ are geologically similar with mountain peaks and flatter valleys as well as coastal terraces. Large rookeries of Adélie Penguins (*Pygoscelis adeliae*) can be found on both islands. For this reason the two localities are commonly visited by tourists. The soils of the study area on Devil Island are very sandy with embedded gravel; vegetation was not existent (Fig. 15). Two study plots were located at the edge of the penguin rookery and two in an area less frequented by penguins (both plot pairs consisting of an influenced area and a non-influenced area approximately 50-100 m distant from the former). Devil Island was studied in 2010. The soils of Paulet Island were very rocky with a thin, very muddy clay matrix and vegetation was also not present Fig. 16). The study plots were located in an area strongly frequented by penguins along a meltwater stream; the anthropogenically non-influenced area was approximately 50 m distant from the influenced area. An additional non-influenced area was also sampled here, on which an exceptional, but extended matt of the algae *Prasiola crispa* grew. Sampling took place on Devil Island in 2011.

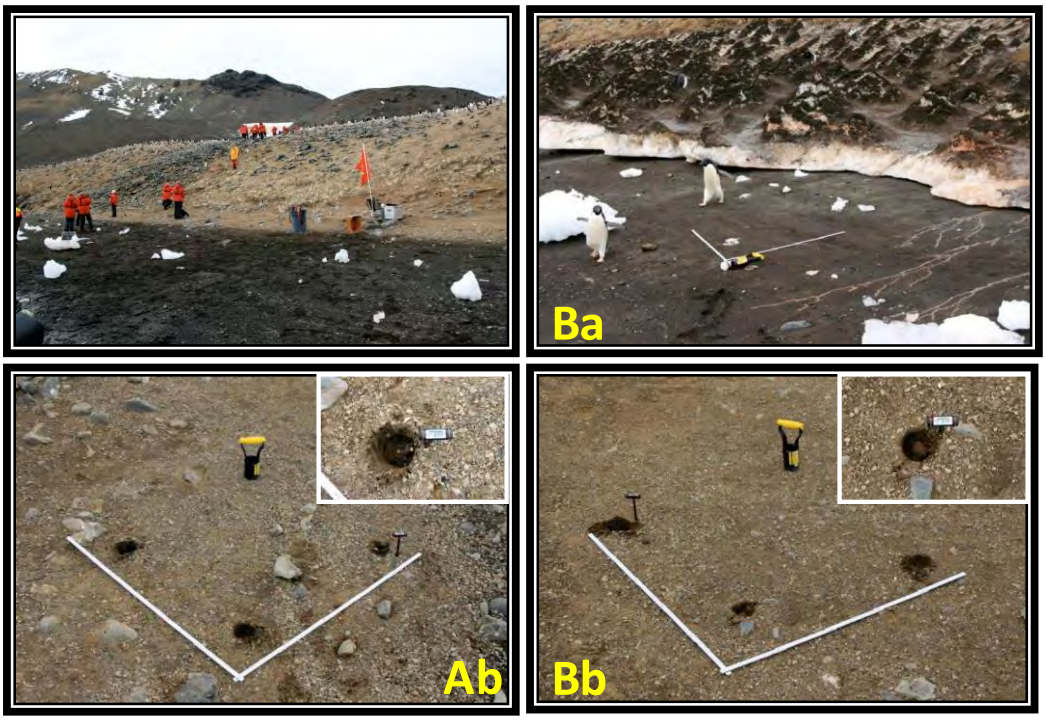


Fig. 15: Sampling plots („areas“) of the locality „Devil Island“ in the Weddell Sea (examples). Figure explanations as in Fig. 7.

Fig. 16: Sampling plots („areas“) of the locality „Paulet Island“ in the Weddell Sea (examples). Figure explanations as in Fig. 7.

The two southernmost study areas were located on the western side of Antarctica Peninsula. The locality „**Neko Harbour**“ is situated on the eastern edge of the bay of the same name. The study sites were located on an ice-free coastal area, which was otherwise surrounded by glaciers and was the site of a Gentoo Penguin (*Pygoscelis papua*) rookery. Next to Whalers Bay, this locality is one of the Antarctic areas most strongly visited by tourists; the anthropogenic influence is caused solely by tourists. The soils in the study areas consisted of larger gravel embedded in clay; the samples were taken from the clay matrix (Fig. 17). The influenced areas were located along the footpath used by tourists; the non-influenced areas approximately 20 m from the influenced areas beyond the footpath (with one exception, which was further separated from the others). The locality was sampled in both study years.

„**Petermann Island**“ is an approximately 1 km long island, lying in the Penola Strait. It generally consists of rock and is largely covered by snow and ice. In coastal areas a few snow- and ice-free spots can be found, on which rookeries of Adélie and Gentoo Penguins (*Pygoscelis adeliae* and *P. papua*) can be found. In the ice-free areas, a few spots of gravelly and non-vegetated sand can be found between larger rocks, from which the samples were taken (Fig. 18). An anthropogenic influence was almost exclusively caused by tourists. Influenced areas were located in the visitor areas near the penguin rookeries; the non-influenced area proximally 20 m distant from the former. The locality was only sampled in 2010.

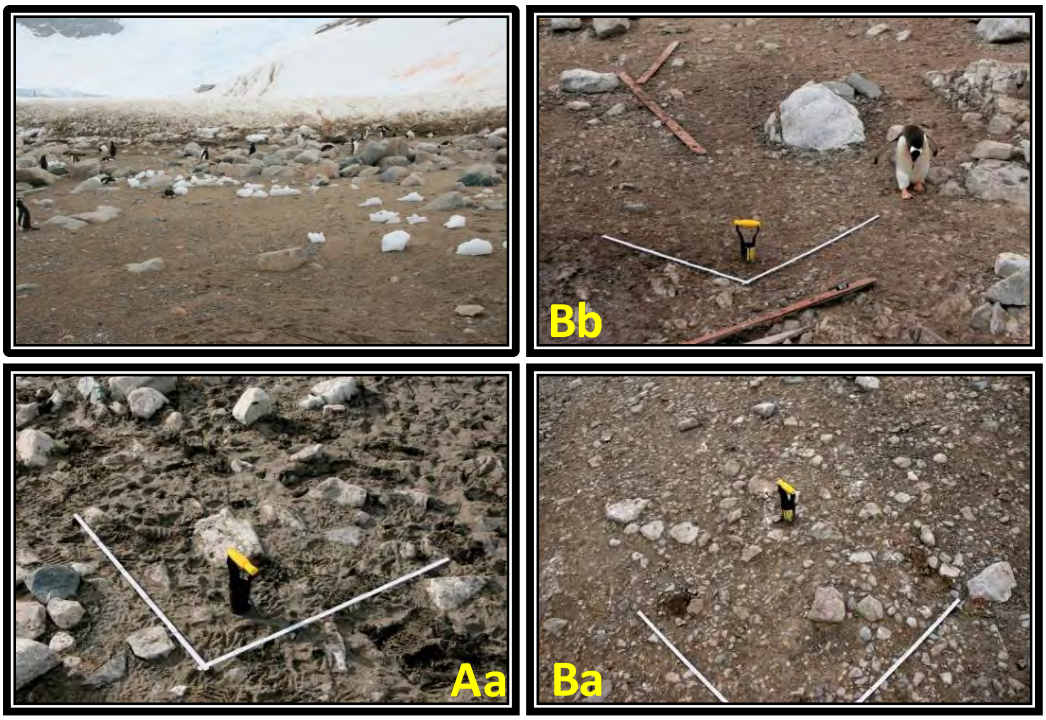


Fig. 17: Sampling plots („areas“) of the locality „Neko Harbour“ on the Antarctic Peninsula (examples). Figure explanations as in Fig. 7.

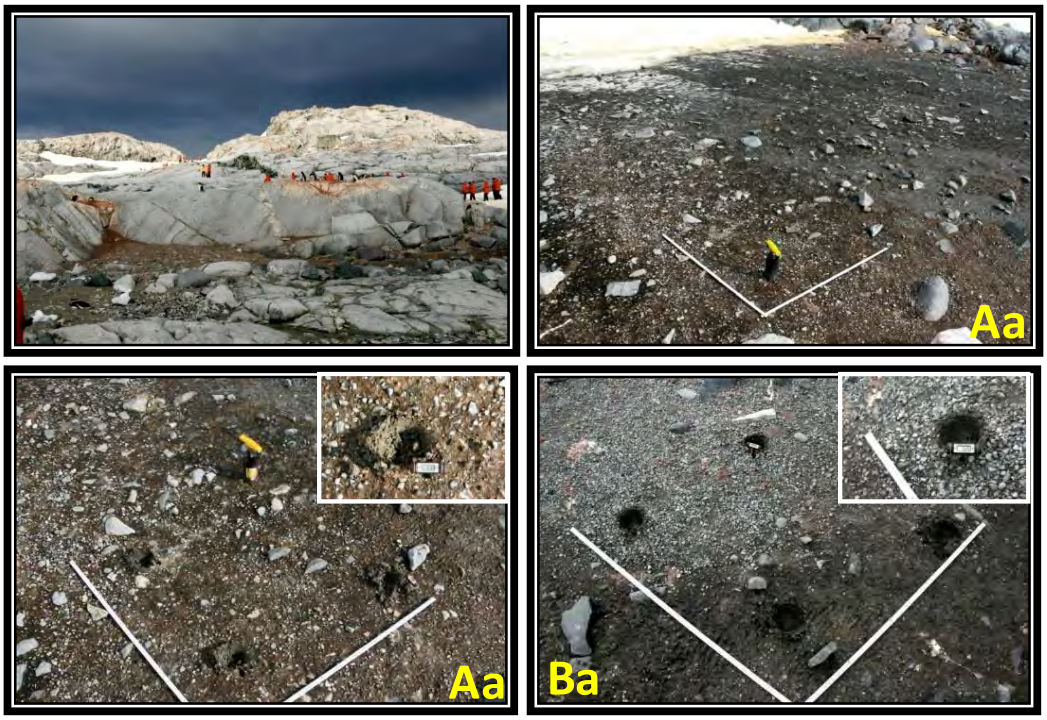


Fig. 18: Sampling plots („areas“) of the locality „Petermann Island“ (examples). Figure explanations as in Fig. 7.

2.1.5 Sampling

Due to the limited amount of time available during the land excursions, not all study plots (= “areas”) could be completely sampled in all localities visited by the cruise ships. In other areas, e.g., on King George Island, the complete sampling programme could be carried out, i.e., in four different localities in the first study year (2010). In the second study year (2011), the localities frequented by tourists were sampled entirely during tours of the MS Hanseatic, mainly

by A. Kertelhein (and additionally by D. Russell). In this year, sampling of single localities took place during multiple ship tours so that the complete sampling program could be carried out. In all sampling sites in both study years, even if only a reduced number of areas were sampled, the total number of four soil cores per area could be taken. A statistical analysis of the collected data was therefore possible despite the resulting unbalanced sampling design. Furthermore, almost all expedition leaders and scientists could obtain the soil cores with the provided soil corer and to similar soil depths (approx. 4 cm). In all sampling sites, the exact measurements of the single samples were recorded, so that all collected data were comparable after data transformation to a standardized sample volume (see below). The soil samples were packaged in commonplace 2 l plastic freezer bags, labeled and double sealed. During the remaining ship tour, the samples were stored at 5-8 °C (e.g., in the ships' flower storeroom) until arrival in Ushuaia, Argentina. The vegetation samples, which are taken separately from the soil samples, were packaged in small paper bags, labeled, sealed and also stored cool and dry. The samples provided by the workgroup of H.-U. Peter from the Fildes Peninsula were taken immediately before their return to Germany, so that special storage of the samples before being transported to Germany was not necessary. The data of the individual sampling occasions are given in Table 2.

Table 2: The localities actually sampled during the austral summers 2010 und 2011, their exact positions, sampling dates, person actually performing the sampling („Sampler") as well as the number of sampling plots (= areas) per locality and the total number of soil cores taken per locality.

Locality	Coordinates	Date	„Sampler"	Nr. of plots	Nr. of samples
Devil Island	63°47'54.0" S 57°17'24.6" W	17.I.2010	A. Kertelhein	4	16
Halfmoon Island	62°35'42.7" S 59°53'54.8" W	19.I.2010	A. Kertelhein	3	12
Halfmoon Island	62°35'43,9" S 59°54'07,7" W	09.II.2010	H.-J. Spitzenberger	6	24
Whalers Bay (Deception Island)	62°58'43.8" S 60°33'24.4" W	19.I.2010	A. Kertelhein	4	16
Whalers Bay (Deception Island)	62°58'43,5" S 60°33'24,5" W	09.II.2010	H.-J. Spitzenberger	6	24
Petermann Island	65°10'29.3" S 64°08'10.7" W	20.I.2010	A. Kertelhein	2	8
Neko Harbour	64°51'45.9" S 62°26'47.5" W	21.I.2010	A. Kertelhein	4	16
Arctowski Station (King George Island)	62°09'32.6" S 58°27'58.1" W	25.I.2010	K. Pütz	6	24
Biologenbucht (King George Island)	62°11'48.3" S 58°59'28.8" W	21.I.2010	H.-U. Peter	6	24
Punta Cristian (King George Island)	62°11'50.7" S 58°56'33.2" W	22.I.2010	H.-U. Peter	6	24
Punta Cristian II (King George Island)	62°11'53.0" S 58°56'47.5" W	12.II.2010	A. Nordt	6	24
Whalers Bay (Deception Island)	62°58'42.96"S 60°33'29.34"W	02.I.2011	A. Kertelhein, M. Steinhof	6	24
Whalers Bay (Deception Island)	62°58'42.96"S 60°33'29.34"W	07.II.2011	D. Russell	2	8
Telefon Bay (Deception Island)	62°55'43.03" S 60°40'48.83"W	02.I.2011	A. Kertelhein	6	24
Neko Harbour	64°50'41.10"S,	03.I.2011	A. Kertelhein, H. Fries	4	16

	62°31'53.46"W				
Neko Harbour	64°50'41.10"S, 62°31'53.46"W	21.I.2011	A. Kertelhein	2	8
Neko Harbour	64°50'41.10"S, 62°31'53.46"W	10.II.2011	D. Russell	2	8
Halfmoon Island	62°35'45.84"S, 59°54'6.84"W	20.I.2011	A. Kertelhein, H. Fries	6	24
Ardley Island (King George Island)	62°12'38.40"S, 58°56'40.62"W	15.I.2011	H.-U. Peter, S. Janowski, A. Nordt	6	24
Paulet Island	63°34'30.36"S, 55°46'59.04"W	06.I.2011	D. Russell	3	12
Hannah Point	62°39'14.94"S, 60°36'39.84"W	07.I.2011	D. Russell	4	16

2.1.6 Transportation of the samples to Germany

Processing of the samples took place in laboratories in Germany, which necessitated transporting the samples from Antarctica to Germany. For the physical sampling in Antarctica as well as the transportation (import) of the samples Germany, the corresponding permits were provided by the German Federal Environmental agency (UBA) as stipulated by the Act implementing the Protocol of Environmental Protection to the Antarctic Treaty of 4 October, 1991.

The individual tours of the cruise ships were chosen so that the individual scientists and expedition leaders ("samplers") could carry "their" samples directly with them during their return flight to Germany. The samples were therefore transported to Germany by the persons taking the samples themselves or their assistants, whereby in both study years the samples were received by personnel of the Senckenberg Museum of Natural History Görlitz at the airports in Frankfurt and/or Leipzig and immediately transported to Görlitz. Only H.-J. Spitzenberger handed over the samples collected by him on 15 February, 2010, to the ship agent Mr. Maximiliano Abadie (Agencia Maritima Internacional) in Ushuaia, Argentina. Unfortunately, long delays in accessing the samples occurred. Since the samples could not be stored cooled, after five weeks the samples were no longer considered to be representative. Consequently, it was abstained from transporting these samples to Germany and they were no longer further processed.

2.2 Laboratory methods

2.2.1 Processing of the soil cores, extraction of soil animals

After arrival of the samples in Görlitz, they were first separated into vegetation and soil samples. The vegetation samples were immediately forwarded to the botanist Dr. V. Otte (SMNG). Since the soil samples were packaged in plastic bags and thus were no longer intact in their original form and structure, they were first visually inspected and the original dimensions (diameter, depth) of each sample quantified based on the fieldwork protocols, in order for sample volume to be determined for each sample. In the case that vegetation was present on a soil sample, this vegetation was separated from the soil, whereby half of the vegetation sample was forwarded to Dr. V. Otte and the other half remained with the sample processing.

The individual soil samples were subsequently weighed, in order to, firstly, further quantify sample size and, secondly, to determine the fresh weight of each sample for the later measurement of soil moisture (see below). Afterwards, each sample (soil and the remaining

vegetation, if present) was divided into two portions for the two methods used for extracting the soil animals from the samples. For the microfauna (Nematoda und Tardigrada) an active wet extraction modified from Baermann (1917) was used, and for the mesofauna (Collembola and the various acarological groups) an active dry extraction modified from MacFadyen (1961). During division of the samples for the two extraction methods, 50 g was used for the Baermann extraction and the remaining sample for the MacFadyen extraction. If the total weight of the sample was less than 100 g, the sample was divided into equal portions with one half subjected to a Baermann extraction and the other half to a MacFadyen extraction. In all cases the weight of each sample was precisely documented.

Table 3: Time period between sampling in the Antarctic study sites and begin of extraction of the soil animals from the samples in Görlitz, Germany.

Locality	Sampling	Sample arrival in Görlitz	Begin sample extraction	Time elapsed after sampling
Devil Island	17 Jan. 2010	28 Jan. 2008	28 Jan. 2010	11 days
Halfmoon Island	19 Jan. 2010	28 Jan. 2008	28 Jan. 2010	9 days
Deception Island	19 Jan. 2010	28 Jan. 2008	28 Jan. 2010	9 days
Petermann Island	20 Jan. 2010	28 Jan. 2008	28 Jan. 2010	8 days
Neko Harbour	21 Jan. 2010	28 Jan. 2008	28 Jan. 2010	7 days
Arctowski Station	25 Jan. 2010	09 Jan. 2010	09 Feb. 2010	15 days
Biologenbucht	21 Jan. 2010	09 Feb. 2010	09 Feb. 2010	19 days
Punta Cristian	22 Jan. 2010	09 Feb. 2010	09 Feb.2 010	18 days
Punta Cristian II	12 Feb. 2010	09 Mar. 2010	09 Mar. 2010	15 days
Whalers Bay	02 Jan. 2011	29 Jan. 2011	29 Jan. 2011	27 days
Whalers Bay	07 Feb. 2011	16 Jan. 2011	16 Feb. 2011	9 days
Telefon Bay	02 Jan. 2011	29 Jan. 2011	29 Jan. 2011	27 days
Neko Harbour	03 Jan. 2011	29 Jan. 2011	29 Jan. 2011	26 days
Neko Harbour	21 Jan. 2011	29 Jan. 2011	29 Jan. 2011	8 days
Neko Harbour	10 Feb. 2011	16 Feb. 2011	16 Feb. 2011	5 days
Halfmoon Island	20 Jan. 2011	29 Jan. 2011	29 Jan. 2011	9 days
Ardley Island	15 Jan. 2011	29 Jan. 2011	29 Jan. 2011	14 days
Paulet Island	06 Feb. 2011	16 Feb. 2011	16 Feb. 2011	10 days
Hannah Point	07 Feb. 2011	16 Feb. 2011	16 Feb. 2011	9 days

To obtain the best results during extraction of soil animals from soil samples, extraction of the animals should take place as soon as possible after obtaining the samples in the field, optimally on the same day or at least within very few days. Due to the long transportation route from the Antarctic study sites to Görlitz, Germany, this was not possible in the present case. Therefore the danger of changes in the faunistic conditions of each sample, e.g., due to mortality, reproduction, predation etc., was possibly increased. Table 3 lists the time period between sampling and animal extraction for each sampling event. Despite the long transportation route, extraction began in most cases - especially in the first study year - relatively soon after sampling. Exceptions occurred in only three localities, all of which were sampled in January 2011. Since the samples were kept cool during storage on board the ships as well as in the airplanes, *larger* changes in these sample are not likely. Nonetheless, the different transportation times of all samples were documented in order to be able to take these long storage times into account during interpretation of the results, if necessary.

The mesofauna (Collembola and Acari) was actively driven from the soil samples in a high-gradient extractor (dry extractor according to Macfadyen 1961). The individual samples were placed in the extractor between a heating and cooling chamber and were thereby exposed to a thermal and moisture gradient. Since soil animals flee in a downward direction from desiccation and heat (negative xero- and thermotaxis combined with positive geotaxis), they were actively driven from the soil samples by the slow heating and drying of the samples from top to bottom. The animals were caught sample specifically in containers partially filled with a fixation and conservation agent (von Törne mixture: 50% isopropanol with 3% glacial acetic acid and 0.3% formalin). Each extraction process ended after approximately 13 days at temperatures of 45 °C (sample upper surface) and 30 °C (sample bottom surface) and total desiccation of the sample (Fig. 19). After ending the extraction, the collected material of each sample was transferred to 70% ethanol and stored for three weeks to ensure conservation of the soil animals.

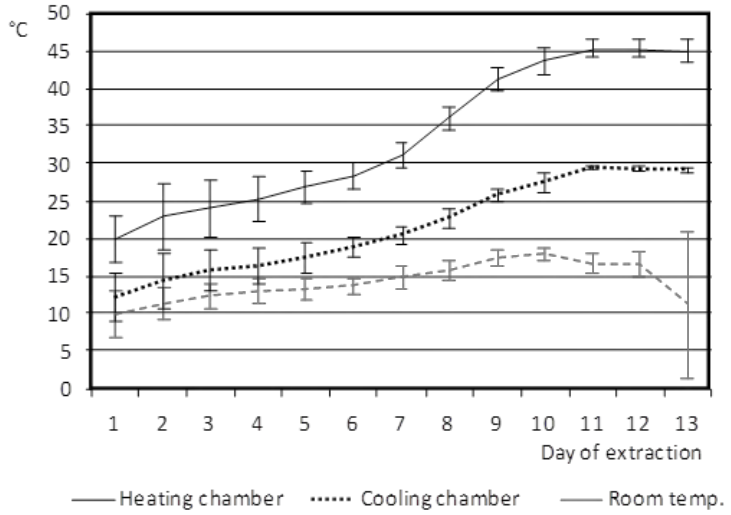


Fig. 19: Course of the temperature gradient (compared to surrounding temperatures) to which the samples were exposed during extraction of the mesofauna.

For the microfauna (Nematoda and Tardigrada), a modified wet extraction according to Bearman (1931) was begun the day of arrival of the samples in Görlitz. For this, each sample was weighed, placed on a milk filter, which in turn was placed on a sieve at the top of a funnel; the funnel ended in a transparent rubber tube secured at its end with a hose clamp. The funnel was then filled with tap water so that lower surface of the soil sample was in contact with the water. The samples then dried slowly from top to bottom at a constant temperature of 20 °C. During this procedure, those soil animals which live in the water film in the soil-pore system (Nematoda, Tardigrada) actively migrated downwards out of the sample (negative xerotaxis and positive geotaxis), sank downwards and were collected in the tube above the hose clamp. To prevent predation and therefore a decimation of animals in the collection tube, the extracted animals were collected daily from the tube and the nematodes and tardigrades present immediately killed with 60 °C water and subsequently conserved in 0.2% triethanolamin-formalin solution (TAF) at 4 °C. Each extraction was ended after five days.

After extraction and conservation of the animals, they were separated from remaining soil (which had dropped into the samples during the extraction process), sorted into the taxonomic

major groups (Collembola, Actinedida, Oribatida, Gamasina, Nematoda and Tardigrada) under the stereomicroscope at maximally 50x magnification and a preliminary sample-specific quantification of these major groups carried out. Processing of the mesofauna and microfauna extraction samples were not performed in parallel, but rather successively. Since the individual densities of the microfauna in individual samples was enormous, despite the smaller extracted sample sizes, the preliminary quantification of the nematodes and tardigrades (in samples with more than 100 individuals) represented merely a rough estimation. The individuals in the samples were precisely counted during the taxonomic determination to species level.

After separation and sorting, the individuals of the major animal groups were transferred to 70% ethanol (nematodes remained in 0.2% TAF solution) and stored sample specifically in small museum collection tubes. For the further taxonomic determination, the Nematoda, Actinedida and Gamasina remained in the Natural History Museum in Görlitz (with Drs. K. Hohberg, D. Russell and A. Christian), and the samples of the other animal groups were sent to the scientists of the other project partners (Table 4). As can be seen from the table, processing of the samples and especially sorting the animals into the major group required approximately five months in each of the study years.

Table 4: Shipping dates of the extracted and sorted soil animals to the taxonomists of the present project.

Animal group	Taxonomist	Date	Samples
Oribatida	A. Bruckner, Vienna	27 Mai, 2010	Complete
Collembola	M. Potoapov, Moscow	24 March, 2010	Sample charge I
		04 June, 2010	Sample charge II
Tardigrada	S. McInnes, Cambridge	07 July, 2010	Sample charge I
		20 Aug., 2010	Sample charge II
Oribatida	A. Bruckner, Vienna	24 Aug., 2011	Complete
Collembola	M. Potoapov, Moscow	15 Oct., 2011	„picked up personally“
Tardigrada	S. McInnes, Cambridge	18 June, 2011	Complete

2.2.2 Determination of the soil organisms

For species determination, microscopic slides of specimens of the respective taxonomic major groups were prepared, often one slide per individual. During the determination of the Collembola and the oribatidid mites, at times large numbers of individuals of single species were sampled. In these cases, the individuals were first separated under the stereomicroscope into morphospecies. The determination of the Collembola and Oribatida otherwise took place with the temporary mount technique. With this method, animals are transferred individual by individual into a half open cavity slide, the cavity slide is filled with the liquid embedding fluid and half covered by a coverslip; the individual specimen can be turned in all directions with a fine needle and observed under the microscope. Among the Collembola, more than half of the individuals were thus embedded in Gisin's and Phoera Liquor and determined to species level. The remaining individuals were quantified and stored in 70% ethanol for future raster electron-microscopic studies. The determination of endemic Antarctic Collembola followed the determination keys and taxonomic revisions of Wise (1967), Massoud & Rapoport (1968), Greenslade (1995) and Deharveng (1981). For Collembola and species not exclusively occurring in Antarctica, the determination keys of Fjellberg (1998), Pomorski (1998), Potapov (2001), Thibaud et al. (2004) as well as Dunger & Schlitt (2011) were used.

A portion of the collected and sorted Oribatida were transferred into lactic acid and warmed on a heat plate (at approximately 40 °C) for a few days to clear and render them transparent for the microscopic determination. Determination took place under the microscope at maximally 600x magnification. The determination of the recorded species followed Hammer (1958) and Wallwork (1962, 1965). The remaining individuals were quantified species specifically. For final storage of all individuals of the Oribatida, they were subsequently transferred to 70% ethanol. Permanent slides were not prepared.

Each individual of the Gamasina was cleared in the glacial acetic acid-glycerin mixture and subsequently mounted in a permanent slide in a gummi-arabicum mixture and determined to species level under a differential interference contrast microscope at 400-1000x magnification. All individuals of the Actinedida were also mounted in permanent slides in a chloral-hydrate gummi-arabicum mixture. The determination of individuals to species level also took place under a differential interference contrast microscope at 400-1000x magnification. The determination followed publications of Strandtmann (1967), Booth (1984), Booth et al. (1985), Usher & Edwards (1986), Kethley (1990), Kethley & Welbourne (2010) as well as very many original species descriptions.

The nematodes were pipetted sample specifically in 0.2% TAF solution onto a large coverslip (mass slide) and counted under a Leica DMI 3000 B inverse microscope at 50x magnification. Subsequently, 100 individuals per sample were determined to species level under an inverse microscope with differential interference contrast at 630-1000x magnification. In samples that contained less than 100 nematodes, all individuals determined. The determination followed publications of Andrassy (1998, 2008), Boström (1995, 1996), Holovachov & Boström (2006), Maslen (1979a), Nedelchev & Peneva (2000, 2007), Peneva et al. (1996), Timm (1971) and further species descriptions. Reference specimens of the Nematoda species per location were embedded in glycerin as permanent slides and sealed with paraffin.

Vascular plants (the two only naturally occurring Antarctic vascular plant species *Deschampsia antarctica* and *Colobanthus quietensis*), lichens and bryophytes were determined to species level wherever possible. For the analyses, however, the mat-forming mosses *Sanionia uncinata* and *Sanionia georgicouncinata* as well as the cushion-forming mosses *Andreaea gainii*, *Andreaea depressinervis* and *Andreaea regularis* were summarized to *Sanionia* spec. and *Andreaea* spec., respectively, in cases where more than one species of these genera were present at a given site. This was done since it is nearly impossible to distinguish the species in the field and haphazard collection of one or the other species does not necessarily translate to true differences in species composition of different collection sites. Among the algae, it was only possible to determine the distinctive and common, often predominant macroalga *Prasiola crispa* to species level. Crusts of microalgae, which were occasionally present, were roughly differentiated into green algae, diatoms and “blue-green algae” (cyanobacteria). Determination of the collected specimens took place under binocular reflecting and transmitting light microscopes (Motic ST 39, Leica M 165 C, Leica DM 2500 P). When necessary, sections of the material were made per hand with razor blades and analyzed microscopically. Lichen compounds, which are highly relevant for determination, were determined by spot tests with aqueous solutions of KOH or NaOCl as well as with alcoholic solution of p-phenylene-diamine. Presence of crystal excretions (in the medulla of *Psoroma tenue*) were demonstrated by polarised light microscopy. The determination followed the literature of Bednarek-Ochyra et al. (2000), Henssen & Renner (1981), Ochyra et al. (2008), Olech (2004) and Øvstedal & Lewis Smith

(2001). In doubtful cases, determination results were confirmed by comparison with herbarium material from the SMNG, which originated especially from the 29th Soviet Antarctic Expedition to King George Island in 1984/85. Specimens recorded in the current project were separated by species and were entered together with the necessary documentation in the Herbarium of the SMNG under the accession number 728.

The vegetation was primarily studied as a biotic habitat factor. For this purpose the parameters "vegetation cover" and "plant community" were also recorded. Vegetation cover was roughly estimated in the following categories:

- 0 no vegetation
- 1 vegetation covered up to 25 % of the sampling area
- 2 vegetation covered up to 50 % of the sampling area
- 3 vegetation covered up to 100 % of the sampling area
- 4 vegetation covered 100 % of the sampling area without gaps

The estimation of vegetation cover took place directly in the field. The photo documentation of the "areas" as well as of the "cores" allowed these estimations to be checked and corrected if necessary by the project botanist, Dr. V. Otte, which also prevented estimation differences among the different persons undertaking the sampling. On the basis of the relevant literature, the vegetation units given in Table 5 were discerned.

Table 5: Plant communities (and their characteristics) identified during the course of the present study in the sampled areas.

	Plant community	Characteristics
1	Prasiola-crispa community	Ornithocoprophilous (Pereira et al. 2007, Schaefer et al. 2007), in the most fertilised sites (Olech 2004)
1a	Deschampsia antarctica – Prasiola crispa community	Near penguin colonies on gentle slopes of moraines or hills, where water runs off with bird faeces (Olech 2004)
1b	Algal crusts	Not further differentiated; in places and areas where higher vegetation is absent
2	Bryum pseudotriquetrum formation	In flooded areas and within meltwater drainage streams (Victoria et al. 2009)
2a	Bryum pseudotriquetrum - Sanionia – association	Habitats subjected to flowing water (Ochyra et al. 2008)
3	*Leptogium puberulum* stands	Barren soil substrates of coarser textures in the more hydric sites (Cannone et al. 2006)
4	Sanionia uncinata formation	A number of associations, partly with *Deschampsia* (next community); in flooded areas (Antarctic swamp) (Victoria et al. 2009, Cannone et al. 2006); wet habitats with impeded drainage (Ochyra et al. 2008)
4a	*Deschampsia* - moss	On coastal plains or gentle slopes with a northern exposure; on gravel and moderately moist soils; in sites with an influx of organic matter (Olech 2004)
4b	*Sanionia georgico-uncinata*	In drier sites on the edges of boggy areas, usually on gravely substrates, along coasts or in valleys between glacial moraines (Olech 2004)
4c	Brachythecium austrosalebrosum association	On wet soils and along meltwater streams (Ochyra et al. 2008)
5	Short moss torf and cushion subformation	The most diverse and disparate moss-dominated communities, typical for fellfields on drier stone substrata (Ochyra et al. 2008)

5a	Andreaea community with *Ceratodon* and lichens, dominance of *Ochrolechia frigida*	On scree slopes covered with boulders (Olech 2004)
5b	Polytrichastrum alpinum formation, with *Ochrolechia frigida*, *Psoroma* etc.	On well drained sites, on rocky substrates (Victoria et al. 2009); tall moss turf on well drained soils and slopes (Ochyra et al. 2008); near the summits of glacial moraines (Olech 2004). Epibryoic crustose lichens in the later stages of succession, or in drier places (Olech 2004)
5c	Chorisodontium aciphyllum stands	Moss turf subformation; on moist, well drained rocky hillsides, forming tall moss banks up to 5500 years old (Ochyra et al. 2008)
5d	Usnea antarctica – U. aurantiacoatra community and Usnea–Andreaea community	In well drained sites (Schäfer et al. 2007); drier and more exposed stands (Cannone et al. 2006), on the summits of predominantly younger moraines (Olech 2004)
6	Species-rich tundra with fruticose lichens	On sites with particularly advantageous, wind-sheltered conditions (Olech 2004)

In a rough summary, the first three communities (1-1b) can be characterized as colonizers of sites with extreme conditions, e.g., which typically show a strong influence of vertebrate animals (trampling and excrement, e.g., from penguins) or where due to climatic conditions no other vegetation can develop. In contrast, all other communities colonize "vegetation-friendly" habitats, whereby the communities 2-4 characterize moist to especially waterlogged sites (geomorphological depressions or plains, meltwater discharge) and the remaining communities dryer habitats (knolls, gravelly slopes).

The assignment to specific plant communities was based on vegetation samples collected in the field additionally to the soil samples, together with the photo documentation of the sampling sites (cores and areas), which allowed the relative abundances of the species to be estimated. In particular, the external samplers were urged to provide - besides the "soil cores" - additional vegetation samples from the "areas" in order to minimise effects of local randomness at the spatial scale of individual cores, thereby documenting as far as possible the entire botanical diversity of the sampling areas detectable in the field. Nevertheless, the vegetation data from the soil cores (from which the soil fauna were obtained) and the additional area samples (from which no soil animals were collected) were complied separately for the statistical analyses, since the smaller area of the soil cores could have a higher relevance for the soil fauna (due to their reduced mobility) than the total botanical character of a specific area, which appears "homogeneous" only in overview.

2.2.3 Soil analyses

For a detailed characterisation of the sampled soil substrates, the following soil parameters were assessed for each individual sample (exceptionally combined for a sampling area if too little substrate was available per sample for an analysis):

- soil temperature
- soil water content (= „soil moisture")
- pH value
- C_{org} (= organic carbon; „mass loss at ignition")
- particle size distribution (= "soil texture")
- C/N ratio

The assessment of these parameters is necessary for the evaluation of differences in species composition, population sizes or distribution of individual species, which can possibly be solely

due to habitat conditions. First after this influence is known, can the comparability of anthropogenically influenced and non-influenced sampling areas be ensured and the biological results evaluated in regard to a possible human impact.

Soil temperatures were measured in the field during sampling with a digital soil thermometer and documented in the field protocol. Usually one measurement was taken per sample, at the very least one measurement per area. For the determination of **soil moisture**, in the laboratory in Görlitz after the completed MacFadyen-extraction, each individual sample was dried at 105 °C and the dry weight subsequently determined. The (gravimetric) soil moisture of each sample was determined from the fresh weight (before extraction of the soil animals: Gew_F) and the dry weight (after extraction of the animals: Gew_T) according to the formula:

$$\text{Soil moisture [\% dry weight]} = (Gew_F - Gew_T * 100) / Gew_F$$

After determination of the soil moisture, each partial sample from the Baermann- and Macfadyen-extraction was frozen and stored at -22 °C for the future soil analyses, since the remaining soil parameters were analyzed first after the soil animals had been sorted from the extracted material. Previous to further measurement of abiotic parameters, individual samples were thawed and subsequently dried again sample-specifically in the drying oven at 105 °C.

pH values were measured according to the specifications of the VDLUFA (1991) as well as DIN 10390. For the measurements, individual samples were passed through a test sieve of 2 mm mesh size in order to disintegrate any soil aggregates. From the sieved soil, 10 g of each sample was weighed, transferred to a Falcon Tube and 25 ml of a 0.1 M KCL-solution added. The samples were subsequently mixed on a shaker machine for one hour. Subsequently, after sedimentation of the soil substrate, the pH values were measured with a freshly calibrated laboratory pH-meter and a Hannah glass electrode.

The soil was also passed through a 2-mm test sieve before measurement of the **organic carbon content** (C_{org}). Subsequently, 5 g of each sample was weighed into small laboratory porcelain bowls, dried overnight in the drying oven at 105 °C, after which the exact dry weight (Gew_T) was determined. The individual samples were then placed in a muffle furnace, incinerated for three hours at 550 °C and subsequently placed in the drying oven overnight at 105 °C to cool. Afterwards, the dry weight after incineration of each sample (Gew_G) was measured. The content on organic carbon from each sample was determined according to the formula:

$$\text{Mass loss after incineration}_{550°C}\text{[\% dry weight]} = (Gew_T - Gew_G * 100) / Gew_T$$

For the determination of the **particle size distribution (soil texture)**, the organic carbon content (C_{org}) was first measured. At C_{org} contents of more than 2%, the organic material in the soil of the respective sample was removed by oxidation with concentrated hydrogen peroxide before analysis of the particle-size distribution. For this, samples were dried as described above in the drying oven at 105 °C and then passed through a 2 mm test sieve, whereby larger organic remains (dried moss particles, dry grass etc.) were removed manually. The samples were subsequently transferred to a 1000 ml beaker, de-ionized water was given until the sample was thoroughly moistened, and then 20-30 ml concentrated H_2O_2 was added. After repeated stirring, the samples were left to stand overnight and then placed in a 40 °C water bath until the H_2O_2 degradation was completed (no further visible production of oxygen bubbles). The sample was subsequently dried at 105 °C. For the final determination of the particle size distribution, the total weight of the individual thoroughly dried sample was

determined and then placed in the uppermost sieve of a sieve-shaker machine. Each individual sample was passed through a test-sieve cascade of the mesh sizes 20 mm, 6.3 mm, 2 mm, 0.63 mm, 0.2 mm and 0.063 mm for 20 minutes. Subsequently, the masses of the soil material remaining in each of these sieves (= particle size fraction) was determined and recorded as the percent of the total weight. For samples containing very little soil substrate (e.g., in the samples from King George Island with much vegetation), mixed samples of the individual samples of one area were produced and only one measurement per area was made.

The analyses of the **contents of soil carbon and nitrogen** were determined spectrometrically according to DIN 10694 in a Vario Pyro Cube analyzer. For this, the sample was dried and passed through a 2-mm test sieve as described above and subsequently lightly crushed in a mortar and pestle in order to disintegrate all aggregates into primary particles (however, without grinding the primary particles). From each sample 5 g of soil was packaged and sealed in a tin capsule and placed in the pyrotechnic spectrometer. With the samples taken in the year 2010, five replicates per individual sample were measured; due to the low variability of the measurements in this year, only two replicates per individual sample were measured from the 2011 samples. In this year outliers (defined as measurements that were more than one standard deviation from the average of all replicates) were discarded and two further replicates measured. The average value of the individual replicates per sample were taken as the result for each individual sample in the statistical analyses. The values obtained in these measurements were subsequently used to determined that **C/N ratios** of each individual sample.

2.3 Zoological data analysis

2.3.1 Univariate statistical analyses

The number of species and the number of individuals of each species in each specific sample represented the raw data, with which all further analyses were performed. For each sample, the individual density of each species was extrapolated and standardized to individuals per 100 cm^3 with the formula:

$$\text{individual densities per 100 cm}^3 = x/n^*100$$

whereby x represents the number of individuals and n the volume of the actual (Baermann or Macfadyen) extracted sample. The volume of each individual sample was determined from the volume of the total sample (calculated from the diameter and depth of the sample, which was documented for each sample in the fieldwork protocol by the person taking the samples) and the portion of the sample (in percent dry weight) used for the particular extraction method. Extrapolation of the individual densities of each species in each sample was necessary in order to standardize the differently sized samples as well as the different sample portions used in the two extraction methods and thus to guarantee the data comparability of all samples as well as all animal groups. Individuals per volume is an unusual unit in modern soil zoology (normally densities are given in individuals per m^2); however, a unit based on volume had to be chosen, since the different sizes of the samples taken in the field (both concerning diameter as well as depth) did not allow a standardization based on surface area.

For the specific animal groups, the individual densities of the respective species were summed to obtain the total densities of the respective group. To obtain total values for the microfauna and mesofauna, the densities of the respective animal groups were added together as well.

Arithmetic averages of the densities of each species as well as each animal group were calculated for each sampling plot (= "area") as well as each locality from the specific samples of the respective area or location.

To determine whether significant differences in densities or species richness existed between localities as well as between anthropogenically influenced and non-influenced areas, the respective data ("total densities" or "species number per sample") were submitted to a **non-parametric variance analysis (ANOVA)**: a modified Friedmann test for multiple observations (= samples) per cell (= area or locality) (Zar 1999). This variance analysis is based on ranked data (instead of absolute values) per plot (or locality) as well as on the χ^2- rather than the F-distribution and can easily handled unbalance sampling designs. First, the variance analyses were performed with the main factor "locality" (combining the influenced and non-influenced areas of the respective locality), in order to determine whether significant differences existed between the densities or species richnesses of the individual localities. A Tukey-like post-hoc test for this non-parametric ANOVA subsequently tested for significant differences between individual localities. For those localities that were sampled in both study years, the data were also submitted to the variance analysis with the main factor "year" in order to test for possible differences between the two study years. For the major part of these analyses, the main factor "area" was tested for the sum of all study sites (localities), with which the statistical significance of possible differences between anthropogenically influenced and non-influenced areas was tested. Since preliminary analyses showed large differences between the study years, the data from 2010 and 2011 were first analyzed separately and then subsequently together. For all of the variance analyses mentioned here, the sum parameters (total densities, species numbers) of each animal group, of the animal groups combined into microfauna and mesofauna as well as of the total fauna were evaluated, provided sufficient individuals in enough samples were present for a variance analysis.

To determine whether significant relationships existed between the soil animal species and habitat parameters, the zoological data were submitted sample specifically to a **Spearmann correlation analysis** (with the software Statistika V 10) together with all abiotic soil data as well as the botanical date (species number per sample, the vegetational cover in percent). The individual densities of each species per sample represented the zoological data basis - provided sufficient individuals of a species were present in many samples; the total densities and species numbers (as a measure of species richness) of the taxonomical major groups were also evaluated. From the resulting correlation data matrix, only highly significant correlations ($P \leq 0.001$) were retained.

Since background habitat parameters (such as soil moisture, soil temperature, content of organic material etc.) can also influence the occurrence of single species as well as total species communities, differences in the influence of these factors can possibly mask anthropogenic impacts on the faunistic species communities. For this reason, **co-variance analyses (ANCOVA)** were carried out, with which the fraction of the total data variance caused by these individual habitat factors (= "co-variables") can be filtered out. For this, a two-way variance analysis was performed with the total densities and species richness of each animal group as well as with densities of each individual species and the two categorical factors "treatment" and "vegetational cover". As covariates, every soil parameter as well as sampling date and the geographical coordinates of the respective study site (the latter two variables as proxies for the factor "locality") were used. The abiotic soil factors were submitted to a Cox transformation and

the zoological data to log(x+1)-transformation previous to the ANCOVAs. Especially for the analyses of the total densities as well as the species richnesses of the major taxonomic groups, at first all covariables were included in the calculations, and subsequently all covariables that only insignificantly accounted for the zoological data variability were removed for the final analysis. Not all preconditions for this parametric statistical procedure were met, e.g. normal distribution of the data (despite data transformation), equal variances of the individual factors (homoscedasticity) or a linear relationship between the covariables. This can lead to erroneous results, which however usually concern Type-II (= ß) errors. This means that true differences may not be recognized as a statistically significant result. On the other hand, if this procedure recognizes results as being statistically significant, then these do indeed reflect true differences (in this case between "treatments", "degree of vegetational cover" or interactions between these two factors). Non-significant results cannot be interpreted as meaning that no differences exist between the tested factors, but only that - due to the data quality - they could not be recognized.

2.3.2 Multivariate statistical analyses

Previous to these analyses, the raw abundance data of the investigated animal groups (number of individuals * 100 cm^{-3}, soil cores of each area averaged) were also logarithmically transformed ($x_i' = \log(x_i + 1)$, with x_i = raw data) to order to reduce the influence of very dominant species on the statistical results. The Bray-Curtis Index was used as a measure of similarity among the assemblages, since this index is "usual" for this type of analysis due to its favorable statistical properties (Clarke & Warwick 2001).

The similarities of the studied animal communities were characterized with **NMDS (Non Metric Multidimensional Scaling**, Clarke & Warwick 2001) to detect influences of the tested factors "locality" and "treatment" (tourist influence yes/no) on community structure. NMDS is a very robust ordination procedure that represents the ranked similarity among objects (here: animal communities) in low-dimensional space. The resulting 2D- or 3D-plots can be interpreted analogous to a geographic map: similar animal assemblages are plotted in close distance to each other and vice versa. NMDS is an unconstrained (free) ordination procedure, meaning that the total data variability is analyzed and represented (as opposed to CAP, see below).

The **PERMANOVA** routine was used to formally test whether the factors "locality" and "treatment" significantly influenced community similarities (Anderson et al. 2008). In analogy to an analysis of variance (ANOVA), the routine permutes community similarity matrices to partition total data variability and calculate significance values for each term in the statistical model (Anderson et al. 2008). In this manner, it is possible to avoid many of the problems classical multivariate procedures (e.g., MANOVA) have with abundance data of biological assemblages, due to non-normal frequency distributions with positive skew, and an overabundance of zero and extreme counts. A type III (partial) model was used with locality as a random and treatment as a fixed factor. We permuted model residuals 999 times.

An important assumption to validly interpret the PERMANOVA output is the absence of significant differences in multivariate dispersion among groups. This is analogous to the homoscedasticity criterion of an ANOVA; in a NMDS plot, such differences can be seen in differently spread clusters of localities. Multivariate homogeneity of the data sets was tested for with the **PERMDISP routine** (Anderson et al. 2008). In essence, this procedure compares the distances of individual localities to the centers of their respective cluster (the group centroids)

and calculates an F-statistic. If the statistic is significant, the PERMANOVA significance values must be interpreted carefully, since they are not statistically reliable. To run PERMDISP, 999 permutations were used.

To isolate the influence of treatment while holding all other factors constant, **CAP (Canonical Analysis of Principal Coordinates**, Anderson et al. 2008) was used. CAP is a guided (constrained) ordination procedure, which "focuses" on the influence of only one selected factor; note the contrast to NMDS, which aims at capturing total variability. CAP is thus a robust permutative analogon to a discriminance analysis: even subtle effects of a factor may be revealed which may be masked by the "noise" of other factors in a NMDS. A cross validation was used check the success of the CAP procedure in correctly identifying groups of objects (here: groups of localities deferring with respect to treatment). CAP models were run with all localities except one (multiple runs with each locality excluded in turn) and the performance of the models in correctly classifying the excluded localities ("leave one out allocation of observations") subsequently calculated. At the best, 100% of all localities are correctly classified. Additionally, a trace statistic was calculated to test for significant effects of the factor under consideration. CAP was run with 999 permutations.

The PERMDISP procedure described above was also used to test for significant differences in the β-diversity among animal assemblages. For this, the presence/absence data and the Jaccard Index (percentage of species restricted to only one assemblage) were used and tested for the factor "treatment" (= anthropogenic influence). As a substantial influence of the factor "locality" was evident from the proceeding analyses, the PERMDISP procedure was run individually for each locality and over all investigated animal groups. All multivariate calculations described here were conducted with the software PRIMER 6.1.12 and the PERMANOVA+ 1.0.2 add-on (PRIMER-E Ltd, Plymouth, UK).

2.4 Assessment of the efficiency of the bootwashing procedure used aboard the MS Hanseatic

On the various land excursion sites of the MS Hanseatic (Tour 1102 in the year 2011), approximately 100 persons (passengers, expedition team and ship crew) went ashore. The passengers received high-quality rubber boots (with a strong sole profile) from Hapag-Lloyd for the land excursions. Both before leaving the ship as well as upon returning, each "land visitor" had to walk through a pan with a disinfectant agent (Fig. 20, left). Members of the ship crew monitored this prevention measure and ensure that it was always abided by. Two "changing cabins" on opposite sides of the ship on the same deck served the passengers as an area to change their footwear as well as storing their rubber boots etc. An apparatus for automatically cleaning and disinfecting the boots was present in one of the cabins ("bootwashing machine"; Fig. 20, middle). Both cabins contained many hand brushes (with a water connection) for manual cleaning of the rubber boots (Fig. 20, right). The use of this cleaning equipment as well as the cleansing of the footwear by the passengers was not monitored by the ship's crew.

Fig. 20: Biosecurity and cleaning measures on board the MS Hanseatic to prevent transfer of biological material between (Sub-)Antarctic localities. Left: Disinfection pan; middle: bootwashing equipment; right: hand brush with water connection.

The investigations described here aimed at assessing the efficiency of the cleaning measures and thus the potential of transferring soil organisms from location to location, e.g. via the rubber boots used during landings. For this, after different land excursions and after use of the cleaning measures by the passengers, the rubber boots were carefully washed again on board the ship ("control cleansing"), the wash water collected and retained for examination of the possibly present soil organisms. It was originally planned to carry out the control cleansing after three land excursions (a Subantarctic island as well as a northern and southern excursion site on the Antarctic Peninsula). In the end, the assessment took place on 1 February, 2011 after visiting Salisbury Plain (South Georgia, Subantarctica) as well as on 7 February, 2011 after the excursion on Deception Island (South Shetland Islands, northern Maritime Antarctica). The originally planned third assessment could not be carried out, since the cruise ship could not anchor at the targeted locality due to the prevailing pack-ice conditions and the remaining travel time after later excursions was insufficient for the necessary sample processing (see the following).

After the land excursions, a random sample of the passengers' rubber boots (10 pairs for each assessment, corresponding to approximately 10% of the boots used during the exclusions) was chosen for the control cleansing. The soles and lower sides of the rubber boots were carefully washed with tap water over a large funnel (55 cm diameter). The wash water (about 300 ml each) was collected and retained in glass beakers, one for each pair of boots. The control cleansing was begun with an airbrush device modified as a type of "mini-high-pressure cleaner". However, in contrast to preliminary tests carried out in Germany before traveling to Antarctica, the pressure produced by this device was not sufficient to clean the deep profiles of the rubber boots, so that in the end the boots were cleaned with a common laboratory squeeze wash bottle.

After being washed out of soils, it is known that soil organisms remained suspended in the liquid used for long periods of time, which would severely hamper the further processing of the samples without further treatment (especially reduction of the volume of wash water). Therefore, the wash water was transferred sample specifically to Baermann funnels (fixed at the narrow end with rubber tubes closed with hose-clamps) directly after the control cleansing, so that possibly occurring soil organisms could settle into the lower areas of the rubber tubes (Fig. 21, left). Before traveling to Antarctica, this procedure was tested in the laboratory (in Görlitz, Germany) to determine the time necessary for and efficiency of soil organisms settling into the rubber tubes. For this test, Baermann funnels were filled with ca. 400 ml tapwater and about 100 individuals of different nematodes species in the funnel fixed on a mechanical shaker (Fig.

21, right). The shaker was subsequently set that the lowest rolling shake setting to simulate ship movement at sea. After 24 hours more than 50% of the added large nematodes species could be found in the lower rubber tube areas, while smaller species remained in suspension.

During the investigations on board the ship, the wash water remained in the Baermann funnels for 4-5 days to allow suspended particles and organisms to settle. During this time the rotating oscillations of the liquid in the funnels caused by the ship's movement were considerably less than in the preliminary tests in the laboratory. Furthermore, the particles suspended in the water samples settled almost completely during this time period, so that a sufficient collection of possibly occurring organisms appeared to be sufficiently ensured.

Subsequently, approximately 25 ml of the wash water containing sedimented material were collected sample specifically in small plastic museum-collection jars, to which ca. 25 ml 96% ethanol were added to each sample, the jars tightly sealed and transported to Görlitz, Germany. There, each sample was carefully inspected under the stereomicroscope at maximally 60x magnification and organisms present in the samples recorded and identified to group level.

Fig. 21: Left: Travel-Baermann equipment used for collecting suspended soil organisms in efficiency-test samples. Right: Equipment used during the preliminary tests.

3. Results

3.1 Abiotic soil parameters

The purpose of the soil analyses was to assure that the different treatment plots (human influenced and non-influenced areas) were comparable in regard to background habitat factors that could also influence the studied soil organisms. Thus the statistical analyses of the measured soil parameters were only performed individually within the single localities.

Soil temperatures measured during sampling expectedly revealed cool substrates. Temperatures were almost always under 10°C, often below 5°C (Fig. 22). Although expected, no north-south gradient among the soil temperatures was observed. The highest temperatures were generally measured at Deception Island. In only a few localities could statistically significant differences between treatments be determined, which overall showed no relation to either influenced or non-influenced areas. These significant differences usually related to absolute differences of around or less than 1°C, which cannot be considered to be biologically relevant for the studied soil organisms. An exception concerned the treatment plots at Whalers Bay (Deception Island) in 2011, where an absolute average difference of 2.5°C was recorded. Although also only limitedly biologically relevant, these differences were due to the high temperature variability among the individual plots, which in this year were at times spatially widely separated.

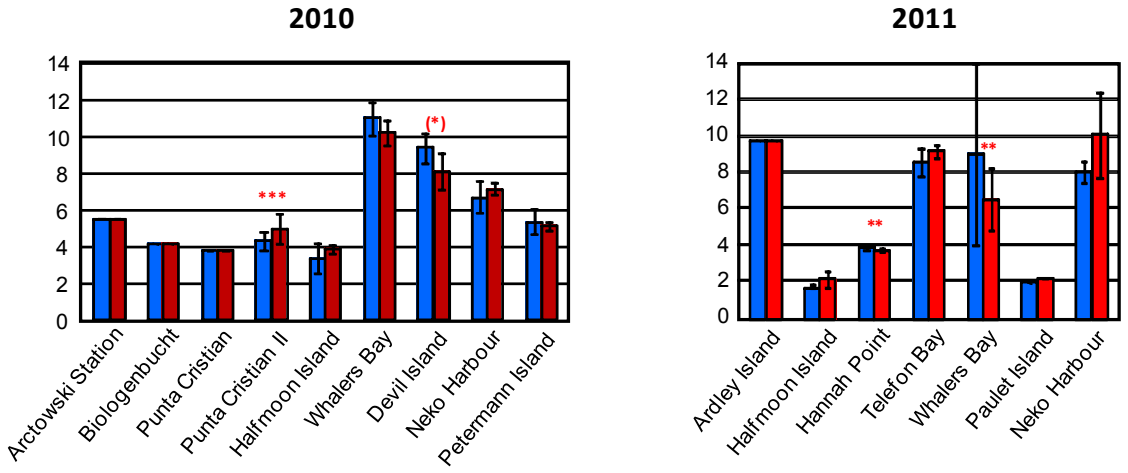


Fig. 22: Soil temperatures measured during the fieldwork in the years 2010 and 2011 in the anthropogenically influenced (blue) and non-influenced (red) areas. Values in °C. Statistical comparisons were only performed between the two area types within the respecitive locality (i.e., not overall). *: $P < 0,05$; **: $P < 0,01$; ***: $P < 0,001$; (*): $0,07 > P > 0,05$.

Soil moistures ranged between 10% and 40% and showed a general north-south gradient of decreasing moistures (Fig. 23). Significant differences between the moistures of the influenced and non-influenced plots could be determined in some localities. However, the absolute differences were almost always within a range of 5%, which again cannot be considered to be biologically relevant. An exception concerned the plots on Halfmoon Island in 2010, where non-influenced plots were on average almost 20% moister, which could influence soil organisms.

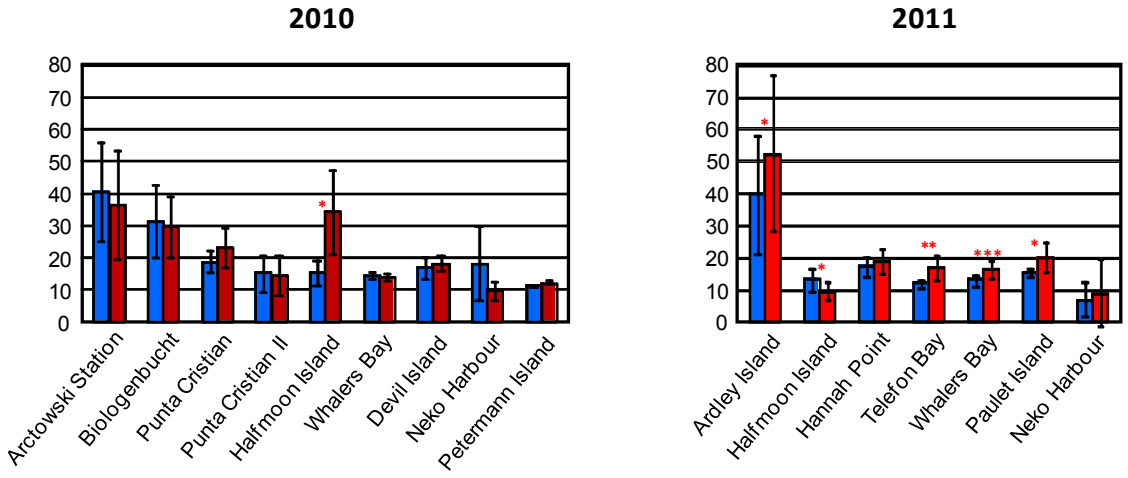


Fig. 23: Average soil moistures of the samples taken in the anthropogenically influenced (blue) and non-influenced (red) areas at the time of sampling in 2010 and 2011. Values in % wet weight. Statistical comparisons as in Fig. 22.

The **pH levels** of the sampled soils range between pH 3.5 and pH 7 (Fig. 24). A north-south gradient of increasing pH was discernible. Especially in the year 2010, many significant differences between treatments within a locality could be observed, which overall also showed no relation to human influence. The absolute differences were again not large (usually within one pH level and never changing the level of acidity of the studied soils), so that again significant differences generally should not translate into a varying influence on the soil organisms.

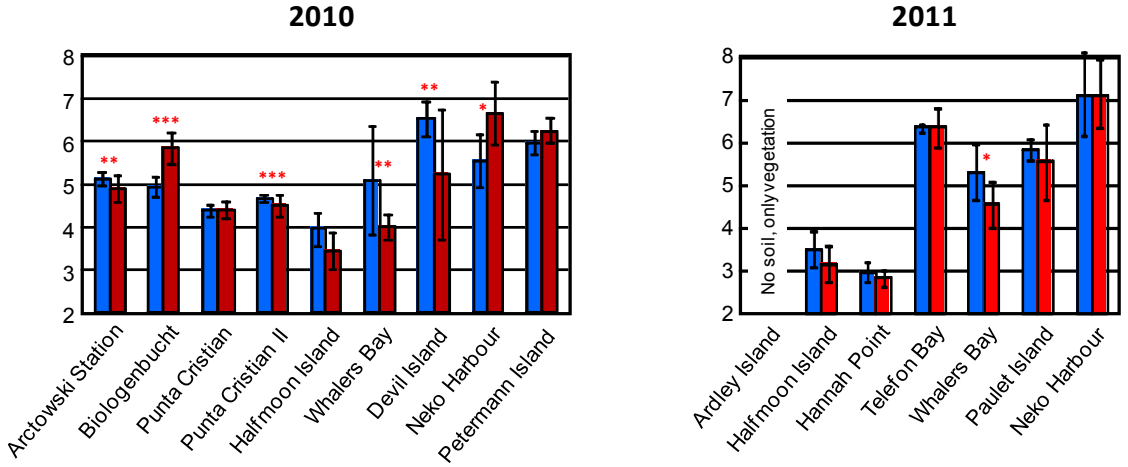


Fig. 24: Average pH values of the samples taken in the anthropogenically influenced (blue) and non-influenced (red) areas at the time of sampling in 2010 and 2011. Statistical comparisons as in Fig. 22.

The sampled soils were generally low in **organic matter**, ranging from 1% to ca. 10% and rarely higher (Fig. 25). An exception represented the soils from Paulet Island, which showed unusually high contents of organic matter (between 15% and > 20%), most likely due to the high input of penguin excrement. In general a north-south gradient of decreasing contents of organic matter could be observed, again with the exception of Paulet Island. Significant average differences between treatments were mostly observed in 2011, which - regarding absolutes values - were again small and most likely represented no differential biological

influence. Here an exception were the soils from Halfmoon Island in 2010, where the non-influenced areas were much richer in organic matter.

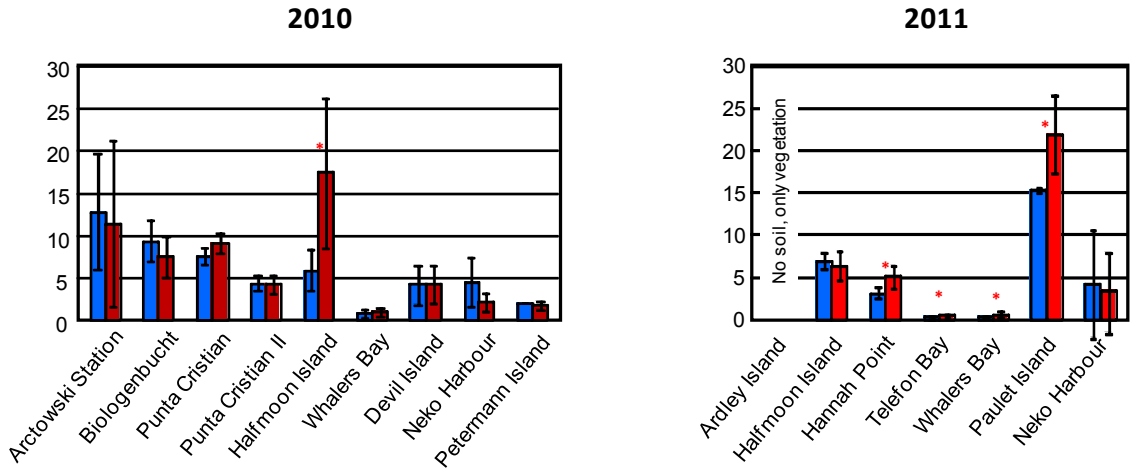


Fig. 25: 2Average contents of soil organic matter of the samples taken in the anthropogenically influenced (blue) and non-influenced (red) areas at the time of sampling in 2010 and 2011. Values in % dry weight (= mass loss at ignition at 550 °C). Statistical comparisons as in Fig. 22.

The **contents of carbon and nitrogen** reflected the generally nutrient poor status of the soils, with soil-C generally ranging between <1% and ca. 6% and soil-N usually being under 0.5% (Fig. 26, top and middle). Exceptions here were again Paulet Island in 2011 as well as Halfmoon Island in 2010, which showed higher nutrient contents. Significant differences were recorded in both years in individual locations, with higher nutrient contents usually in non-influenced areas. Absolute differences were, however, once again relatively low and therefore also biologically not relevant. However, the absolute nutrient level differences in the non-influenced areas in Paulet Island and Halfmoon Island were larger than in the influenced areas. The **C/N-ratios** reflected primarily the low levels of soil carbon and ranged between values of 4 and ca. 10, which can all be considered as reflecting high qualities of organic material (Fig. 26, bottom). The measured levels were generally lower in 2011 than 2010. A north-south gradient of decreasing C/N-ratios could be observed, most likely reflecting decreasing C- contents. Significant differences between influenced and non-influenced areas were rarely discerned; when differences were significant they were again at low absolute levels and thus also not considered to be biologically relevant.

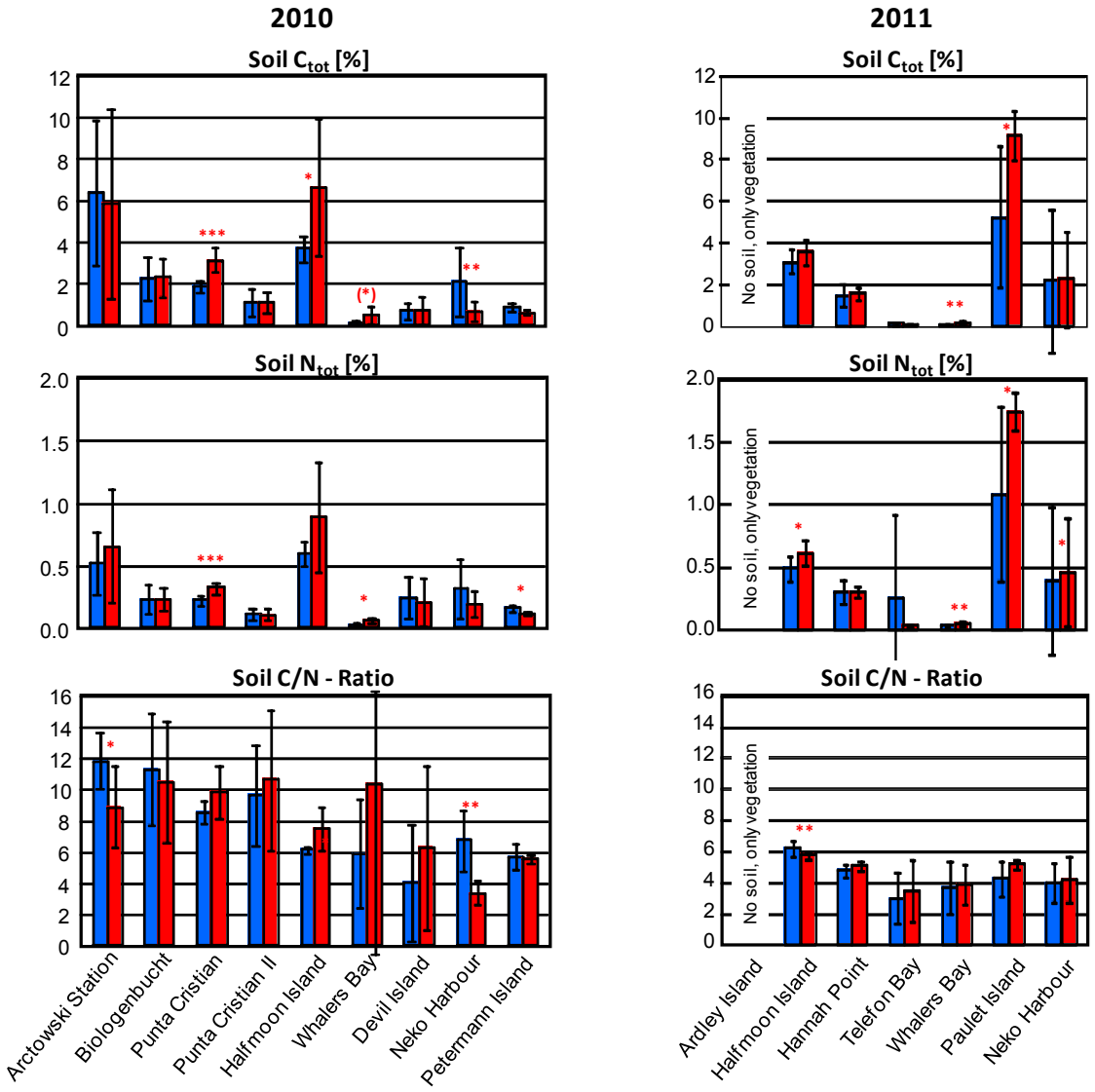


Fig. 26: Average values of various parameters of the soil organic matter in the samples taken in the anthropogenically influenced (blue) and non-influenced (red) areas at the time of sampling in 2010 and 2011. Statistical comparisons as in Fig. 22.

The **soil textures** of all sites were generally sandy and gravelly. Only rarely could differences in the clay/silt content of the soils (which could influence especially the microfauna) be recorded between treatments, and these were again all at low absolute levels. Although statistically significant differences of certain particle sizes could be ascertained between treatments (Fig. 27), more important differences would be between gravel or sand in total. These total differences were rarely above 10%, which also cannot be considered to cause a large differential impact on soil organisms. An exception could possibly be the soils of Halfmoon Island, where those of the non-influenced areas were generally sandier compared to the more gravelly soils of influenced areas.

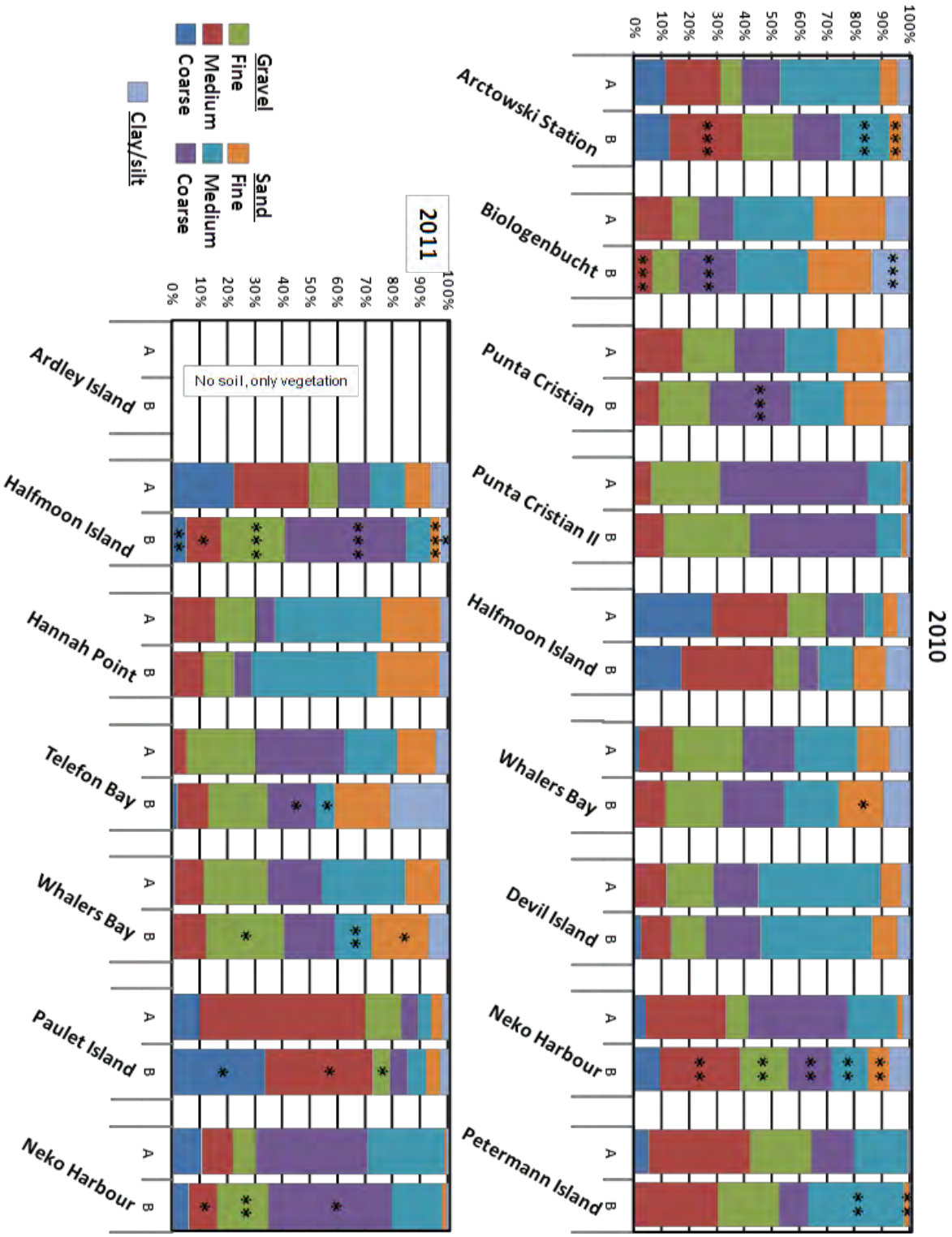


Fig. 27: Average particle size distributions (soil texture) of the samples taken in the anthropogenically influenced ("A") and non-influenced ("B") areas at the time of sampling in 2010 and 2011. Statistical comparisons as in Fig. 22.

3.2 Vegetation

The current project primarily concerns soil animals. Since the heterotrophic fauna is dependent on the autotrophic biomass production, the vegetation of the study sites was also recorded. The – mainly cryptogamic – vegetation was therefore primarily of interest as a biotic habitat factor of the fauna. From the point of view of habitat function, the parameters "degree of vegetational cover" and occurring "plant community" were also documented. Vegetational cover is potentially relevant not only for above-ground biomass (production), but also for a possibly tempering influence of the vegetation on the microclimate, both of which may influence the zoocoenoses. The plant community can be considered to be an integrative long-term indicator of the sum of the local ecological site conditions. Different plant communities indicate with a high probability differing ecological site conditions.

Vegetation (apart from occasionally documented microscopic soil algae) was found in a total of eight of the 13 localities (in 11 of the 18 sampling events). No vegetation was present in the study areas on Devil Island, Petermann Island, Telefon Bay, Whalers Bay (in the year 2011) and Neko Harbour; slight, non-determinable green growth was present in Whalers Bay (single areas in the year 2011) according to the photographic documentation. In two cases (Paulet Island, Halfmoon Island in 2011), pure stands of the thallophytic green alga *Prasiola crispa* were present. Higher vegetation (bryophytes/lichens/vascular plants) was only found in seven localities (at 9 sampling events). Of these 9 sampling events, only insufficient additional vegetation samples were collected at five sampling events (i.e., single moss cushions without any documentation from Halfmoon Island 2010 and Whalers Bay 2010; no or incomplete samples from the "areas" in Arktowski Station and Ardley Island), so that conclusions concerning anthropogenic influences on botanic diversity are very problematic at these localities. Also from the Biologenbucht, the botanical "area" (= additional) samples were not complete. Likewise, no "area" samples were available from Hannah Point; nonetheless, based on the photo-documentation, the plant association in this case truly appears to consist only of the two species recorded with the soil samples (see below). Therefore, if Biologenbucht and Hannah Point are included, four localities may be considered to be sufficiently documented botanically. All are situated on King George Island.

The recorded vegetation consisted primarily of cryptogamous species. Altogether, two species of vascular plants, 24 lichen species, 19 mosses, 3 liverworts and one macroalga were recorded. All of these species were found on the South Shetland Islands (King George Island, Ardley Island), very few also on Halfmoon Island and in Whalers Bay (Deception Island). All species were previously known from the South Shetland Islands. The detailed results are given in Appendix 2.

The species recorded in the current project are given in their systematic position in the following:

Tracheophyta (Vascular Plants)

Colobanthus quietensis (Kunth) Bartl.
Deschampsia antarctica E.Desv.

Bryophyta (Mosses)

Andreaea depressinervis Cardot
Andreaea gainii Cardot
Andreaea regularis Müll. Hal.
Bartramia patens Brid.

Brachythecium austrosalebrosum (Müll.Hal.) Kindb.
Bryum pseudotriquetrum (Hedw.) P.Gaertn. et al.
Ceratodon purpureus (Hedw.) Brid.
Chorisodontium aciphyllum (Hook.f. & Wilson) Broth.
Didymodon brachyphyllus (Sull.) R.H.Zander
Ditrichum ditrichoideum (Cardot) Ochyra
Polytrichum alpinum Hedw.
Polytrichum juniperinum Hedw.
Sanionia georgicouncinata (Müll.Hal.) Ochyra & Hedenäs
Sanionia uncinata (Hedw.) Loeske
Syntrichia filaris (Müll.Hal.) R.H.Zander
Syntrichia magellanica (Mont.) R.H.Zander
Syntrichia saxicola (Cardot) R.H.Zander
Warnstorfia fontinaliopsis (Müll. Hal.) Ochyra
Warnstorfia sarmentosa (Wahlenb.) Hedenäs

Marchantiophyta (Liverworts)

Barbilophozia hatcheri (A. Evans) Loeske
Cephaloziella varians (Gottsche) Steph.
Lophozia excisa (Dicks.) Dumort.

Lichenes (Lichens)

Bacidia tuberculata Darb.
Cladonia gracilis (L.) Willd.
Cladonia pyxidata (L.) Hoffm.
Cladonia sarmentosa (Hook. f. & Taylor) C. W. Dodge
Cystocoleus ebeneus (Dillwyn) Thwaites
Himantormia lugubris (Hue) I.M.Lamb
Lecanora polytropa (Ehrh. ex Hoffm.) Rabenh.
Lepraria cacuminum (A.Massal.) Lohtander
Lepraria straminea Vain.
Leptogium puberulum Hue
Massalongia carnosa (Dicks.) Körb.
Massalongia intricata Øvstedal
Ochrolechia frigia (Sw.) Lynge
Pannaria caespitosa P.M.Jørg.
Placopsis contortuplicata I.M.Lamb
Placopsis parellina (Nyl.) I.M.Lamb
Psoroma hypnorum (Vahl) Grey
Psoroma tenue Henssen
Rhizocarpon geographicum (L.) DC.
Rinodina olivaceobrunnea C.W.Dodge & G.E.Baker
Sphaerophorus globosus (Huds.) Vain.
Stereocaulon alpinum Laurer
Usnea antarctica Du Rietz
Usnea aurantiacoatra (Jacq.) Bory

Algae

Prasiola crispa (Lightfoot) Kützing

The two recorded vascular plants represent the only two species occurring naturally in Antarctica, while among the lichens and bryophytes (cryptogams) only a very low proportion of the species known from Antarctica or the specific studied localities were determined. The Antarctic cryptogamic flora is comparatively species-rich, with 380 lichen taxa known from all of Antarctica (Øvstedal & Lewis Smith 2001) and 252 species from King George Island alone (Olech 2004) or 174 species from the Fildes Peninsula (Peter et al. 2008). Thus, the present study could record barely 10% of the lichen species already known from King George Island. Ochyra

et al. (2008) mention 113 moss taxa occurring in Antarctica, 87 of which in the South Shetland Islands (and 40 from the Fildes Peninsula: Peter et al. 2008). Less than a quarter of the moss species known from the South Shetland Islands were thus documented in the present study. Of the 27 liverwort species occurring in Antarctica and 18 in the South Shetland Islands (Bednarek-Ochyra et al. 2000), only 1/9 of the Antarctic and 1/6 of the species from South Shetland Island were found in this study. On the one hand, this low proportion is due to the fact that many cryptogamic species are inhabitants of saxicolous (= rock) habitats, which were not investigated in this study. Therefore, especially in the more strongly touristically frequented locations, a large portion of the actual touristic influence on the vegetation was potentially not recognized in the present study. On the other hand, the individual density of many occurring species is very low and, therefore, also the probability of their detection when not specifically searched for.

In contrast to the soil samples, where highly comparable samples of defined soil volume were obtained according to a specific sampling scheme, the comparability of the botanical samples collected by different persons – and under time pressure – is not guaranteed. The botanical collections were furthermore incomplete in the majority of cases. The soil samples, which were collected according to scheme optimised for the detection of the soil fauna, are severely affected by chance regarding the documentation of botanical diversity and, in most cases, apparently only reflect the botanic diversity of the study plots (areas) to a limited degree. This is due to the small radius of a soil sample compared to the dimension of a plant individual and – at this spatial scale - the inherent natural inhomogeneity of individual plant distribution. The soil samples did not always reflect this inhomogeneity of the plant communities.

The sums and average values of the botanical community parameters within the different localities are given in Table 6.

For this reason, additional vegetation samples ("area samples") were to be collected. In the Biologenbucht in the study year 2010, these additional area samples indeed documented many additional species' records with the cushion-building mosses compared to the soil samples. Among the other taxonomical groups, the results of the area and soil samples were comparable. Striking differences in floristic diversity between the human influenced and non-influenced areas could not be detected. A minimal shift from pleurocarp to acrocarp mosses in the influenced areas cannot be excluded, but also not statistically substantiated. The information value of the data is further reduced by the fact that additional area samples were only provided for two of the three non-influenced areas, at the expense of a complete documentation of the acrocarp mosses.

On the one hand, the additional area samples of Punta Cristian I contributed appreciably to the documentation of the complete species inventory, in that particularly obvious, but possibly infrequent species (fruticose lichens, specific mosses) could be documented additionally to the soil samples. On the other hand, the inconspicuous liverworts were recorded more by the soil samples, most likely due to unconscious "collection" of these taxa as a consequence of the more spatially oriented soil sampling. In total, the recorded species richness in the influenced and non-influenced areas was exactly the same. Concerning the minimal differences in species composition, no clear tendencies could be determined due to the variability of the results.

Table 6: Floristic community parameters of the study plots (=areas) of the different investigated localities. Sums (species number) and average values (vegetational cover and plant society). Vegetational cover in categories: 0 = no vegetation, 1 = cover up to 25%, 2 = 25-50%, 3 = 50-<100%, 4 = 100%. For the plant societies, see Table 5 (Materials and Methods).

		Locality:	Arctowski Station	Biologenbucht	Punta Cristian	Punta Cristian II	Ardley Island	Halfmoon Island	Halfmoon Island	Hannah Point	Telefon Bay	Whalers Bay	Whalers Bay	Paulet Island	Devil Island	Neko Harbour	Neko Harbour	Petermann Island
		Area	2010	2010	2010	2010	2011	2010	2011	2011	2011	2010	2011	2011	2010	2010	2011	2010
Number of species	influ-enced	Aa	3	1	4	3	5	?	?	2	0	0	0	0	0	0	0	0
		Ab	5	4	7	3	2	?	?	2	0	0	0		0	0	0	
		Ac	1	6	7	4	5		?		0		0				0	
		Ad											0				0	
	non-influ-enced	Ba	2	6	5	3	2	?	?	2	0	?	?		0	0	0	0
		Bb	2	4	6	9	6		?	2	0	1	?	1	0	0	0	
		Bc	3	3	6	3	7		?		0		0				0	
		Bd											0				0	
Vegetational cover	influ-enced	Aa	3.3	3.8	1.8	2.5	3.8	3	2	4	0	0	0	0	0	0	0	0
		Ab	3	3.3	2.8	3.5	4	1.8	2	2.5	0	0	0		0	0	0	
		Ac	0.3	3.3	3.3	1.8	4		1		0		0				0	
		Ad											0				0	
	non-influ-enced	Ba	3.5	4	3	3.3	4	1	1	3	0	1.8	1	0	0	0	0	0
		Bb	4	3.8	3.3	3.3	3.8		1	2.8	0	1.8	1	4	0	0	0	
		Bc	3.5	2.5	3	3.3	4		2		0		0				0	
		Bd											0				0	
Plant society	influ-enced	Aa	4a	4b	5	4b	4b	1	1	1a						1b		1a
		Ab	4a	4b	5	4b	4b	1	1	1a								
		Ac	4a	4b	5	5b	4b	1	1									
		Ad																
	non-influ-enced	Ba	4a	4c	5a	4b	4c	5	1	1a		2						
		Bb	4a	4	5	5b	4b	5	1	1a		2		1				
		Bc	4a	2a	5d	4b	4b	5	1									
		Bd																

The area samples of the second locality in Punta Cristian appreciably improved the determination of the diversity of the sampling areas compared to the soil samples, thereby contributing to the documentation of the total diversity. By these means, the differences in species diversity between the influenced and non-influenced areas were quite striking. A higher diversity of crustose lichens and acrocarpous mosses could be shown in the non-influenced areas.

No area samples were provided from Arktowski Station. The botanical data were therefore provided only by the soil samples. As the photo-documentation of the locality showed, the actually occurring diversity was thus only partially reflected. For instance, *Deschampsia antarctica*, which was not recorded in the soil samples of one of the study plots, actually did occur and the same plant community developed there as in the locality in general. Based on

the soil samples, the documented floristic differences therefore result from small-scale inhomogeneities within the study areas (plots) und do not allow a general evaluation of differences between "treatments" (anthropogenically influenced/non-influenced).

The additional area collections on Ardley Island contributed relatively little to recording species additional to those of the soil samples. They at least showed that the acrocarpous mosses did actually occur to a certain extent in the non-influenced areas, although they were much more weakly represented in the soil samples of these areas compared to those of the influenced areas. The lacking area samples from the influenced areas nonetheless renders an evaluation problematic. Irrespective of the deficient area samples, a higher diversity was documented in the influenced areas, which resulted from the occurrence of many more acrocarpous mosses. It cannot be ruled out that trampling in this plant community (Sanionia community) created open soil areas, in which these moss species could compete against otherwise cover-building species. In contrast, the apparently trampling sensitive fruticose lichens were somewhat more represented in the non-influenced areas. Primarily different site conditions in the influenced and non-influenced areas are, however, possible causes. The areas are also not homogeneous within themselves, fruticose lichens occur in only one plot of both the influenced and non-influenced areas. This was also shown by the photo-documentation. The conditions within the individual areas thus strongly affect the general results, which could therefore be stochastic.

Only one area sample (without further information) was provided from Whalers Bay on Deception Island, which contained the acrocarpous moss *Bryum pseudotriquetrum*. The acrocarpous moss *Ceratodon purpureus* could also be recorded in single soil samples from anthropogenically non-influenced areas. The photo-documentation sporadically showed the thallophytic green alga *Prasiola crispa* in the influenced areas. A robust assessment of a touristic impact on the vegetation is not possible based on this data, even if a promotion of the algae by human trampling to the detriment of mosses cannot be ruled out.

From Halfmoon Island also only one area sample (also without further details) was collected, which contained dead mosses (*Polytrichum alpinum*, *Sanionia* spec.) encrusted with algae, the green algae *Prasiola crispa*, further indeterminable green algae and diatoms. Further moss and lichen samples were apparently lost during transportation from Antarctica. The photo-documentation showed a strong growth of *P. crispa* in the anthropogenically influenced areas, and in the non-influenced area (only one plot) a sparse vegetation of single cushions of acrocarpous mosses. Based on this data situation, robust conclusions concerning an anthropogenic impact are also not possible, although a promotion of the algae to the detriment of the mosses also seems plausible in this location.

The soil samples documented no differences between the species composition of the human influenced and non-influenced areas of Hannah Point. Additional area samples were lacking, but according to the photo-documentation the plant community does indeed consist of only the two determined species (see Appendix 2). However, the photo-documentation does show striking differences in the vegetation cover, with reduced vascular plants compared to algae in the influenced areas.

Against the background of this data basis, only limited conclusions concerning the impacts of tourism on the vegetation are possible. In localities more heavily frequented by tourists and bearing higher vegetation, the thallophytic green alga *Prasiola crispa* appears to be promoted at the expense of the remaining species. This is in accord with the observation that *P. crispa*

also builds stands in natural habitats, where due to a strong influence of vertebrates the growth of other macrophytes is no longer possible (Longton 1988). However, due to the poor documentation of the study areas on Halfmoon Island 2010 and Whalers Bay 2010, such conclusions are uncertain in these localities. Nonetheless, the photo-documentation from Hannah Point in 2011 shows impressively how *Deschampsia* virtually disappears among mats of *Prasiola* in the trampled areas, while in the non-trampled areas *Deschampsia* predominates.

At the three remaining, sufficiently documented and species-rich localities, the results are not consistent. In Biologenbucht and Punta Cristian I clear differences in diversity of influenced and non-influenced areas were not detectable. On the other hand, in Punta Cristian II, botanical diversity was clearly higher in the non-influenced areas, which was particularly the case for lichens and acrocarpous mosses. Whether this is truly a result of an anthropogenic influence in this locality (whose intensity may be lower than in the other two localities) or whether other factors play a role (e.g., primarily unequal site conditions between influenced and non-influenced areas in this locality) cannot be determined with certainty. For instance, conversely, in Ardley Island (despite the lack of area samples from non-influenced plots of this locality), a higher diversity of acrocarpous mosses was documented in the influenced areas. Indeed, an impact on fruticose lichens by trampling cannot be excluded in this locality, but also not verified with certainty.

The low number of sites that were more or less sufficiently documented with botanical samples restricts the evaluation all the more, since the diversity of the observed plant communities is relatively high compared with the number of localities (fairly specific vegetation at each site, resulting in a lack of replicated sampling plots of the same plant communities at different sites). Those localities intensively frequented by tourists naturally harbour a relatively low soil-borne botanical diversity, also in areas not affected by tourists. Since low species diversity is usually connected with high individual numbers of single species, trampling will hardly lead to a loss of species in those conenoses, but abundance and cover may decrease, as shown above in the example *Prasiola crispa* / *Deschampsia antarctica*. In total, an anthropogenic impact on botanical diversity cannot be excluded. Generally, a lack of statistical evidence for botanical differences between influenced and non-influenced areas does not necessarily mean that such differences were truly absent, but only that they cannot be shown with the available data.

No species were recorded that had not been previously known to occur in Antarctica. In those sites harbouring presumed neozoans in the soil fauna, either no (Neko Harbour) or little (Whalers Bay) higher vegetation occurred. An introduction of non-native plant species was thus not proven in the current study. With the methods used in the present study, the detection probability of introduced species occurring in low densities is limited. Therefore, the lack of evidence for an introduction of plant species in the sampling sites cannot be considered to be evidence that an introduction can be ruled out. Especially cryptogamic plants are generally considered to be able to disperse well (e.g., Kappen & Straka 1988). Many of the determined taxa show large distributional areas throughout the world; a typical pattern is a bipolar distribution including the Arctic as well as the Antarctic.

3.3 General faunistic parameters

In the 327 samples taken during the two study years (164 in each year; one sample missing in 2010), a total of more than 320,000 individuals from all animal groups were determined. The most individuals (> 255,000 individuals) were found among the Nematoda, with Tardigrada

(over 37,000 individuals) and Collembola (> 25,000 individuals) representing the next most frequent animal group. A total of 98 species could be identified. Again the largest number of species (40) were found within the Nematoda, with Actinedida (25 species) and Tardigrada as well as Collembola (14 and 11 species, respectively) being the next most species-rich groups.

In both years highly significant differences between the total faunas of the studied localities were found both in species richness as well as in densities (for results of the statistical analyses, see Appendix 5, Table A5-1). These strong differences between localities existed throughout all animal groups. Significantly decreasing individual densities from the most northern to the most southern localities was especially apparent among the microfauna (Fig. 28, top), with the exception of very high abundances of Nematoda in Paulet Island in 2011. A north-south density gradient was not as obvious among the mesofauna, although the highest densities were found among the localities of the South Shetland Islands (Fig. 28, bottom). Decreasing species richness from the northern to the southern localities was found among all groups (Fig. 29.

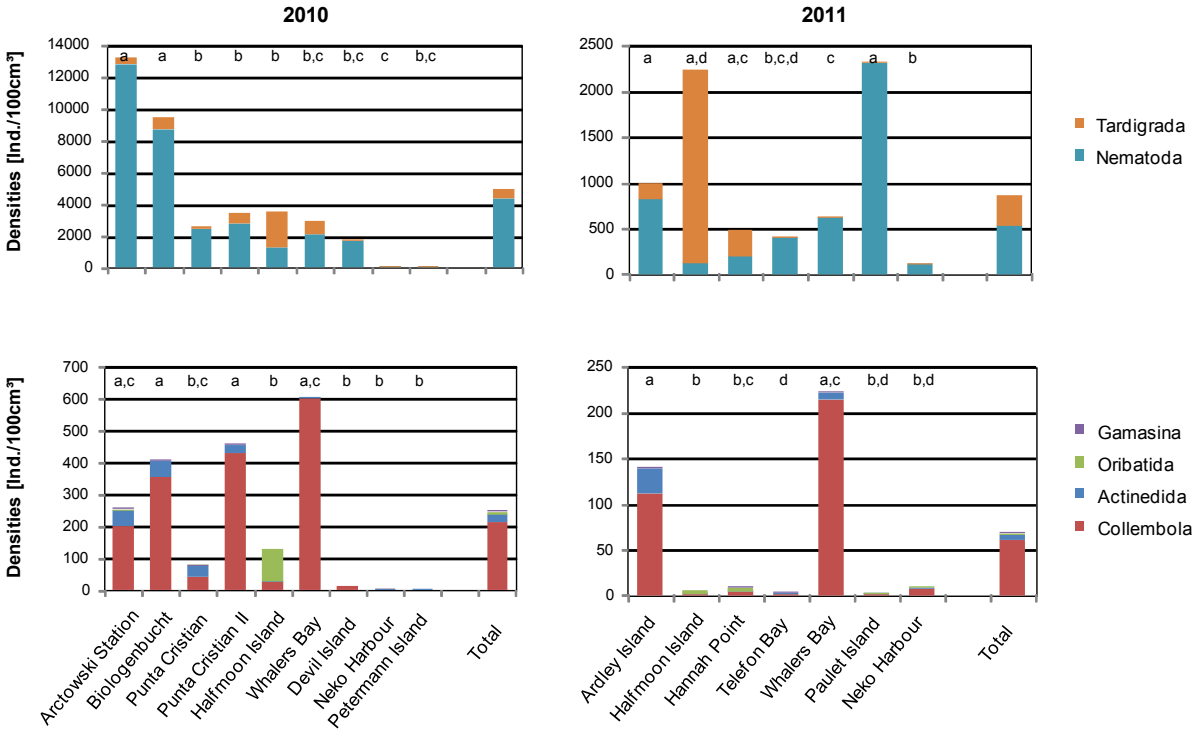


Fig. 28: Total densities (in individuals per 100 cm^3 substrate) of the microfauna (above) and mesofauna (below) in the different localities in 2010 and 2011. Different letters denote significant differences in densities (= the densities of localities with the same letter were statistically *not* different from each other). Please note the different scales of the y-axis of both follow groups.

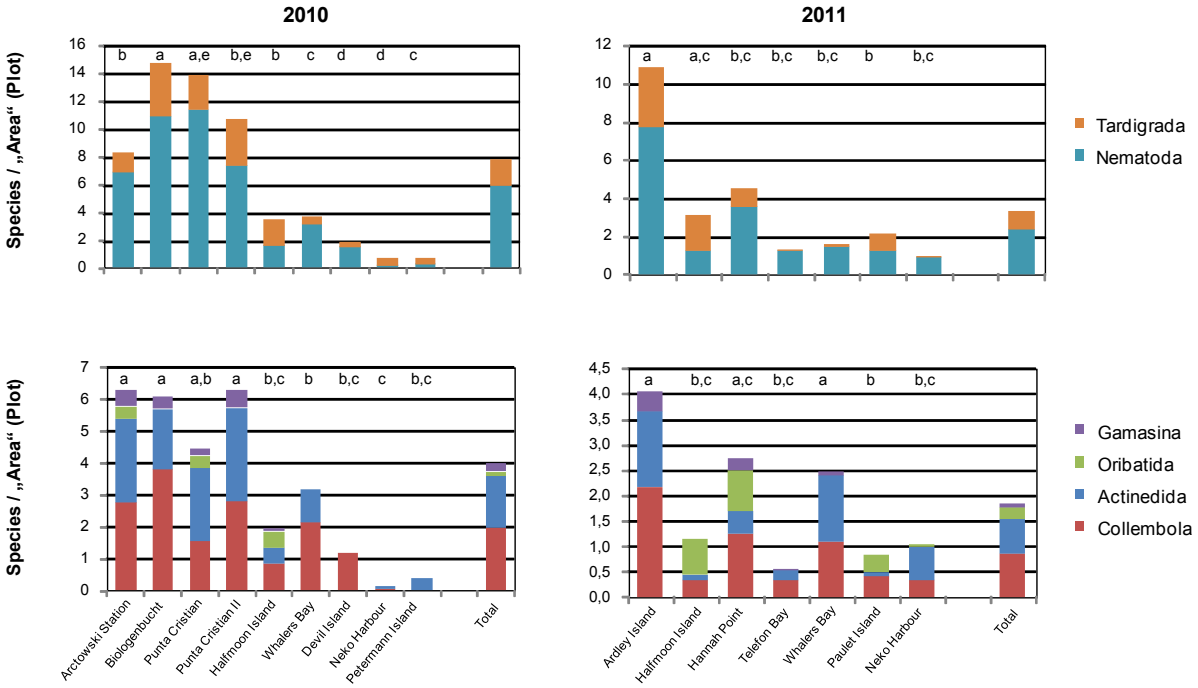


Fig. 29: Species richness (as the average number species per area) of the microfauna (above) and mesofauna (below) recorded in the different localities in 2010 and 2011. Different letters denote significant differences in species richness (= the number of species in localities with the same letter were statistically *not* different from each other). Please note the different scales of the y-axis of both follow groups.

There seemed to be a large year-to-year difference, with over 250,000 individuals and 88 species registered in 2010 compared to the >65,000 individuals and 71 species in 2011. However, this is partly due to the fact that different localities were sampled in the different study years, thus (also) again representing a locality difference. A direct year-to-year comparison can only be undertaken in the three localities sampled in both years (Whalers Bay, Halfmoon Island and Neko Harbour). In these sites, significantly higher densities (total fauna) were found in these localities in 2010 (Appendix 5, Table A5-2), but not regarding the number of species. The difference was most pronounced in the microfaunal groups (analyzed together), both in the higher densities as well as species richness (Appendix 5, Table A5-2) in 2010. Whereby both microfauna groups showed significantly higher densities in 2010, only Tardigrada showed a significantly higher species richness in this year compared to 2011. Only in Whalers Bay did Nematoda show a higher species richness in 2010. In contrast, the mesofauna showed together no significant differences in density or species richness between the two years. Only Collembola showed significantly larger densities in the year 2010. Although more individuals as well as species were found in 2010 in most mesofaunal groups (the exception being Actinedida, where higher densities and more species were found in the three localities in 2011), none of these differences were statistically significant.

Although differences between the human influenced and non-influenced areas (“treatment effect”) were apparent in individual animal groups (Appendix 5, Table 5A-3, s. also results of the individual groups below), these were usually not registered when these taxa were combined at higher taxonomic/ecological levels. For instance, significantly higher densities in the non-influenced areas were only registered among the mesofauna, which was stronger in 2010 (Fig. 30). Taken together, the microfauna showed no significant differences between

influenced and the non-influenced areas, although higher densities were usually registered in the non-influenced sites. On the other hand, only the microfauna showed significant differences in species richness, with slightly more species found in the influenced sites (Fig. 31). However, this was only true in 2010 and these differences were no longer statistically significant when both years were analyzed together. The mesofauna showed no significant treatment effect regarding species richness. When the total fauna was analyzed together, no significant effect of human influence could be ascertained in either total densities or total species richness (Appendix 5, Table A5-3). Thus, treatment affects found in individual animal groups were often no longer found when these taxa were grouped together; in other words the effects found in individual groups were not additive.

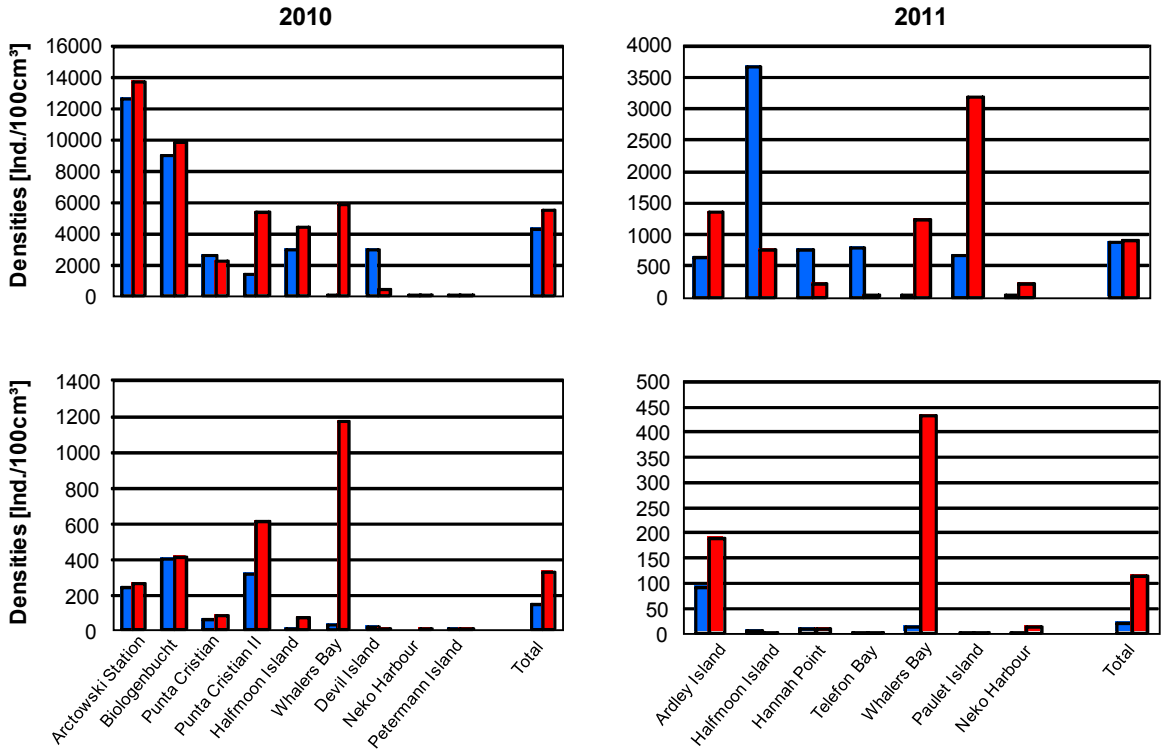


Fig. 30: Total densities (in individuals per 100 cm^3 substrate) of the microfauna (above) and mesofauna (below) registered in the anthropogenically influenced (blue) and non-influenced (red) areas in both study years.

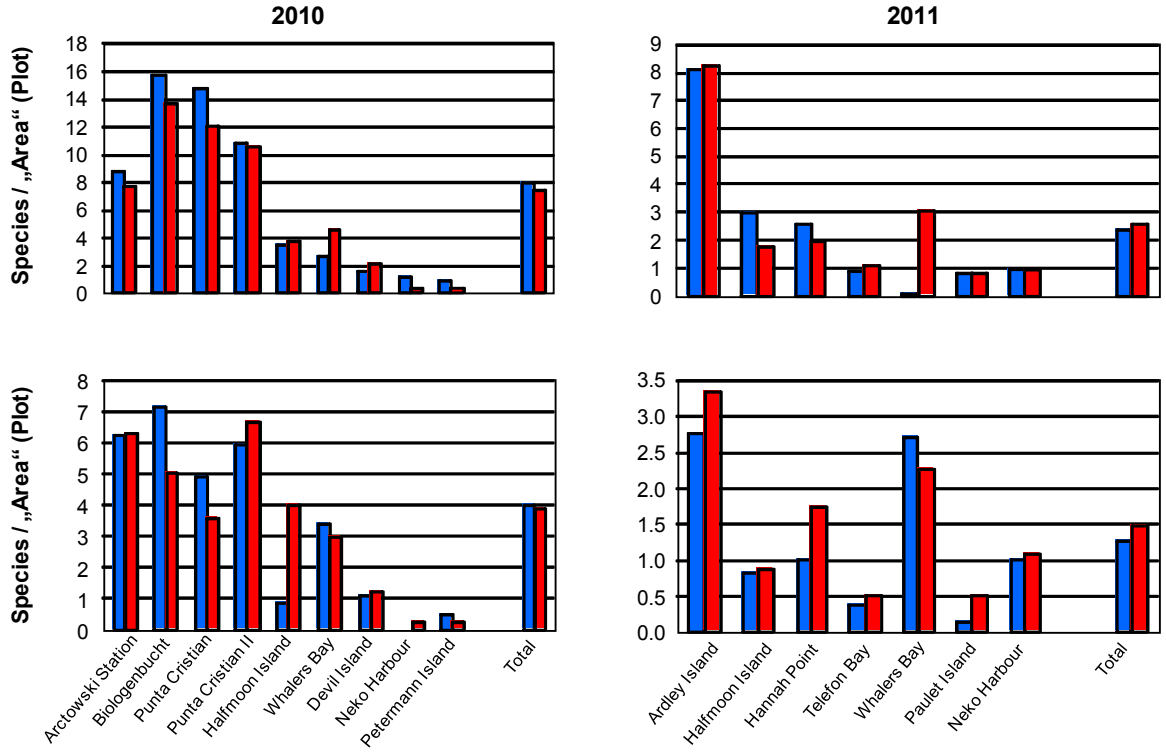


Fig. 31: Species richness (as average number species per area) of the microfauna (above) and mesofauna (below) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in both study years.

3.4 Mesofauna (Microarthropods)

3.4.1 Collembola (Springtails)

In both polar and alpine environments, Collembola (springtails) are among the most abundant terrestrial invertebrates. According to Hogg and Stevens (2002), to date, roughly 15 species of springtails have been recorded from the Antarctic continent. However, this number is obviously underestimated and the true number of species recorded in Maritime Antarctica, Western and Eastern Antarctica (excluding the Subantarctic) amounts to approximately 25. The exact estimation depends on the current taxonomic understanding of the particular species, consideration of single records of species in the Antarctic, etc.

The earliest collections of Collembola in Antarctica were made on August Island near the coast of the Antarctic Peninsula in 1898 (Willem 1901). In the first half of the 20th century, several European taxonomists dealt with Antarctic collections (Wahlgren 1906, Carpenter 1907, Salmon 1949, 1962 and other publications). In subsequent publications (about 200 altogether), various aspects of both morphological and molecular taxonomy, biogeography, distribution and ecology of Antarctic Collembola were considered (see the faunistical and taxonomical reviews of Wise 1967, 1971 etc. and Greenslade 1995, 2010 etc. as well as molecular and methodological approaches by Stevens et al. 2005, Stevens & Hogg 2006, Sinclair 2001, 2006 and others). During the last decade, several papers on the ecology and history of the distribution of Antarctic Collembola were published (Toricelli et al. 2010, Caruso et al. 2009). The maritime Antarctic and Victory Land (Eastern Antarctica) have received the most study,

due to the high concentration of Antarctic research stations and ASPAs (Antarctic Special Protected Areas) in these two areas.

Collembola of the maritime Antarctic, and particularly of the South Shetland Islands, were studied intensively during the last hundred years (Fig. 32; the sites studied in the present project are given in Fig. 33). Several species were described from the maritime Antarctic (e.g., Willem 1901, Wahlgren 1906, Carpenter 1907, Weiner 1980, Greenslade 1995). The last annotated list of collembolan species of the South Shetland Islands was compiled by Greenslade (2010).

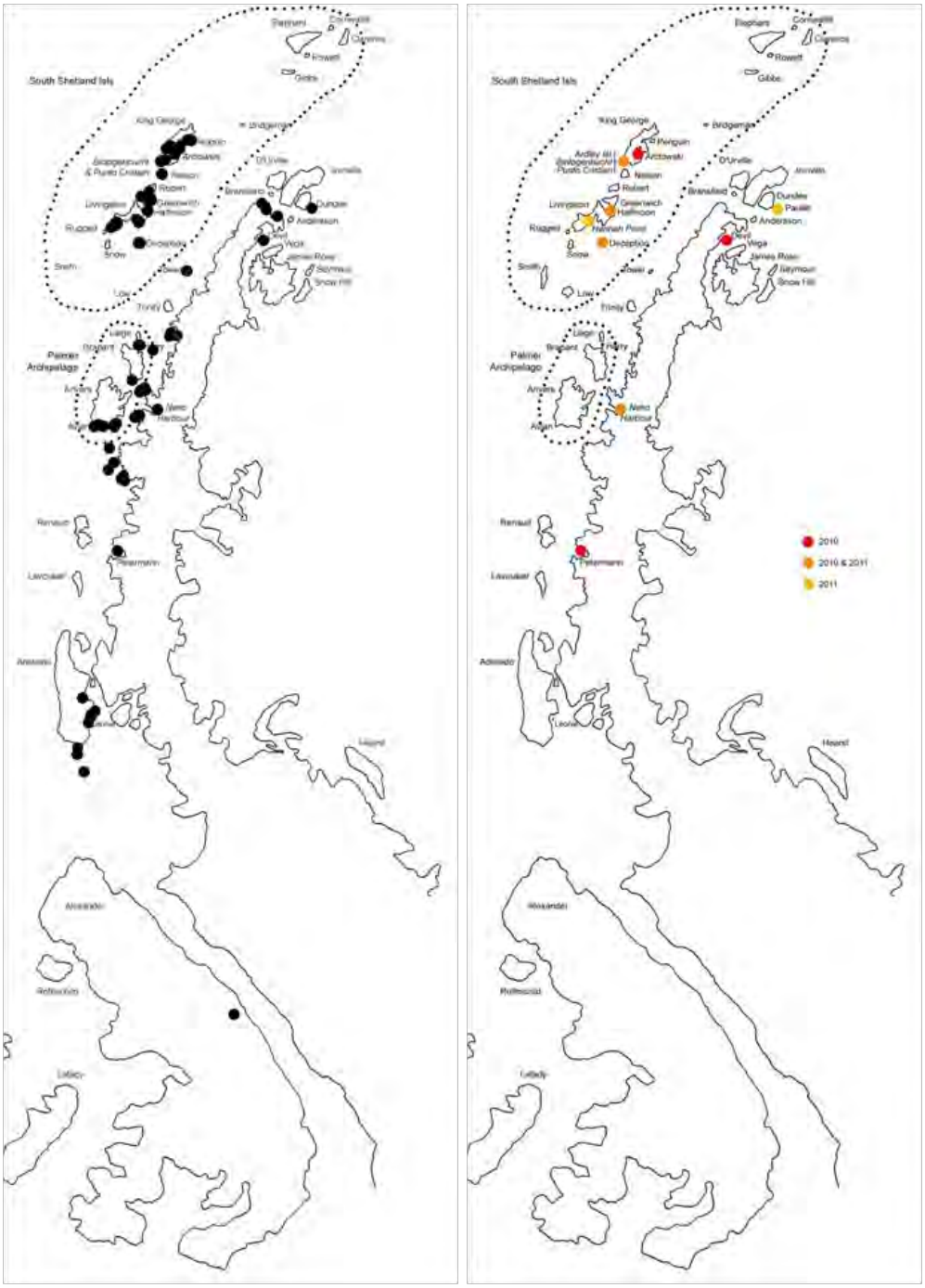


Fig. 32 (left): Locations in the maritime Antarctic of published records of occurrence of collembolan species (including localities studied in the current project).

Fig. 33 (right): The locations studied in the current project. Years of sampling are shown with different colours.

According to Greenslade (1995, 2010) the following species have been reliably recorded from the South Shetland Islands: *Hypogastrura viatica, Tullbergia mixta, Protaphorura fimata, Friesea grisea, Friesea woyciechowskii, Cryptopygus antarcticus antarcticus, Cryptopygus badasa, Cryptopygus caecus, Folsomia candida, Archisotoma brucei* and *Folsomotoma octooculata*. According to literature data, other parts of the maritime Antarctic (Antarctic Peninsula, South Orkney Islands and others) have not shown records of any other species. The species *Hypogastrura antarctica* Salmon, 1962 (= *H. viatica*), *Tillieria penai* Weiner & Najt, 1994 (= *T. mixta*), *Achorutoides antarcticus* Willem, 1901 (= *F. grisea*), *Cryptopygus crassus* Carpenter, 1907 (= *C. antarcticus*), *Cryptopygus nanjiensis* Yue & Tamura, 2001 (= *C. antarcticus*) are considered as junior synonyms of species of the main list. In this list, *H. viatica, P. fimata* and *F. candida* are exotic, or, in another terminology, species non-native to the Antarctic (Greenslade, 1995, 2010).

3.4.1.1 General community parameters

Of the Collembola a total of 25,750 individuals were identified in the present project; 19,299 individuals in 2010 and 6,451 individuals in 2011. This translated into total densities between zero and approx. 600 individuals per 100 cm^3 substrate, with an average collembolan density of about 150 individuals per 100 cm^3 substrate. At the large scale of the maritime Antarctic, the density of Collembola was significantly determined mainly by locality (Fig. 34; for results of the statistical analyses, see Appendix 5, Table A5-1). Localities in and around King George Island (Arctowski, Biologenbucht, Punta Cristian and Ardley Island) and Whalers Bay of Deception Island showed the highest densities.

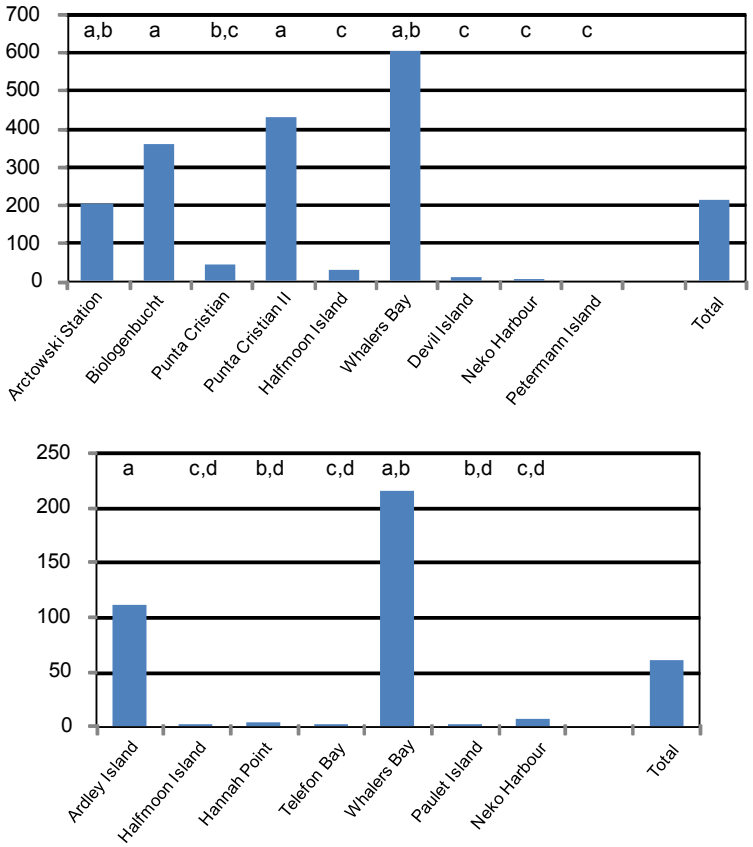


Fig. 34: Total collembolan densities (in individuals per 100 cm^3 substrate) recorded in the various studied localities in

2010 (above) and 2011 (below). Different letters denote significant differences (= localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

Almost three times as many individuals were recorded in the year 2010 than in 2011. This was also true in those localities that were sampled in both years (Whalers Bay on Deception Island, Halfmoon Island near Livingston Island and Neko Harbour on the Antarctic Peninsula), where the densities of these localities together were almost 30% higher in 2010 (Fig. 35; Appendix 5, Table A5-2). This was particularly true in Whalers Bay and Neko Harbour. In Halfmoon Island, on the other hand, in 2010 collembolan densities were found that were almost in order of magnitude higher than in 2011. However, this was due to high individual numbers in single samples, so that high sample-to-sample variability caused the results within Halfmoon Island to be statistically not significant.

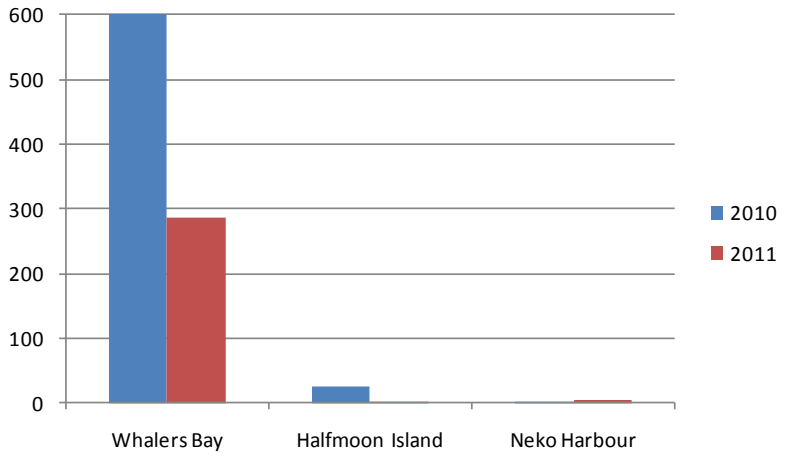


Fig. 35: Total collembolan densities (in individuals per 100 cm^3 substrate) recorded in 2010 and 2011 in the localities studied in both years.

Among other factors, 17 parameters were analyzed (correlation analyses) as possibly relevantly influencing the maritime Antarctic collembolan communities found here, one biotic parameter (vegetational cover) and 16 abiotic parameters (mostly soil factors). Two of parameters, sampling date and latitude, were directly related with locality. Local abundances were apparently considerably affected by latitude, whereby sampling sites at lower latitudes (further south) correlated negatively with density (Appendix 4, Table A4-1), indicating greater abundances the further north the sampling site was (Fig. 36, left). Vegetational cover and collembolan population parameters interacted even more strongly: correlation coefficients were large in both study years (0.554 in 2010 and 0.439 in 2011; Appendix 4, Table A4-1), indicating that the denser the vegetation cover was, the more individual-rich were the collembolan communities. Among abiotic factors, soil texture (the proportions of grain sizes of gravel, sand and silt/clay) showed the least influence on community parameters. Correlations with soil texture were generally not significant or low. Total densities correlated significantly and negatively only with coarser grained material (Appendix 4, Table A4-1). Other parameters appeared to exert a more important influence. Soil moisture apparently relevantly influenced collembolan community abundances, with correlation coefficients as high as 0.499 (in 2011). Several parameters can be grouped and thus considered together, for instance organic matter,

carbon content, nitrogen content and C/N ratio, which all characterize the nutrient status of the soils. The first four characteristics are positively correlated with each other and all correlated negatively with pH. Total densities correlated rarely with these factors of organic material, only showing larger total abundances related to higher qualities of organic material (= C/N-ratio) in 2010. Collembolan densities were apparently affected negatively by pH in 2010, which may reflect the fact that more acidic soils showed lower contents of organic material.

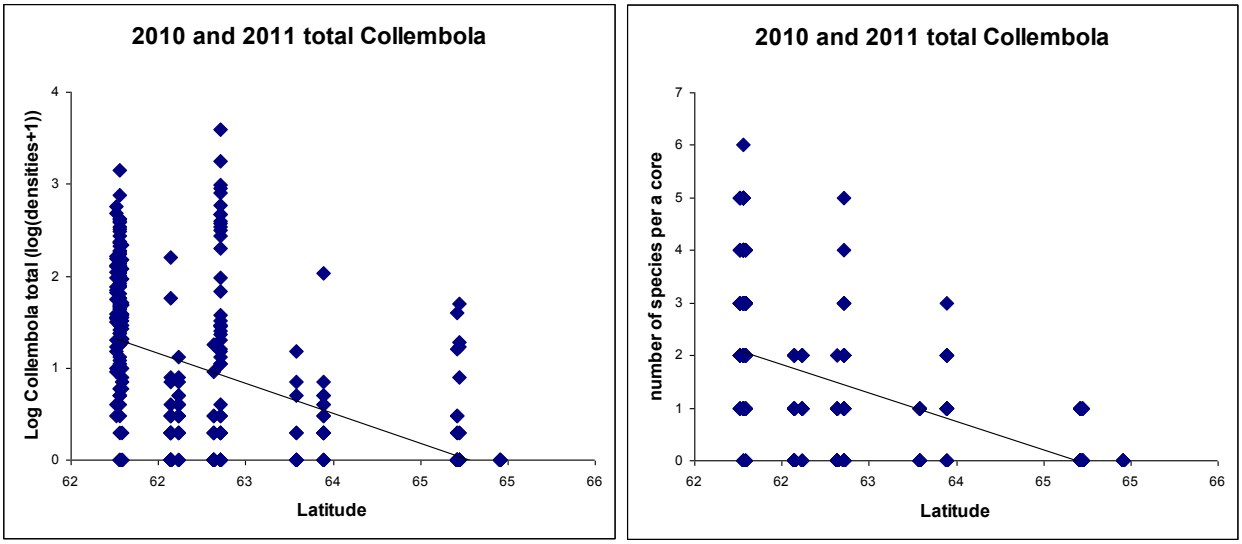


Fig. 36: Correlation between total density (left) and species richness (right) and latitude.

When all study sites were analyzed together, collembolan densities were significantly higher in the anthropogenically *non*-influenced areas than in the influenced areas (Fig. 37; Appendix 5, Table A5-3). These effects were particularly strong at Whalers Bay (Deception Island).

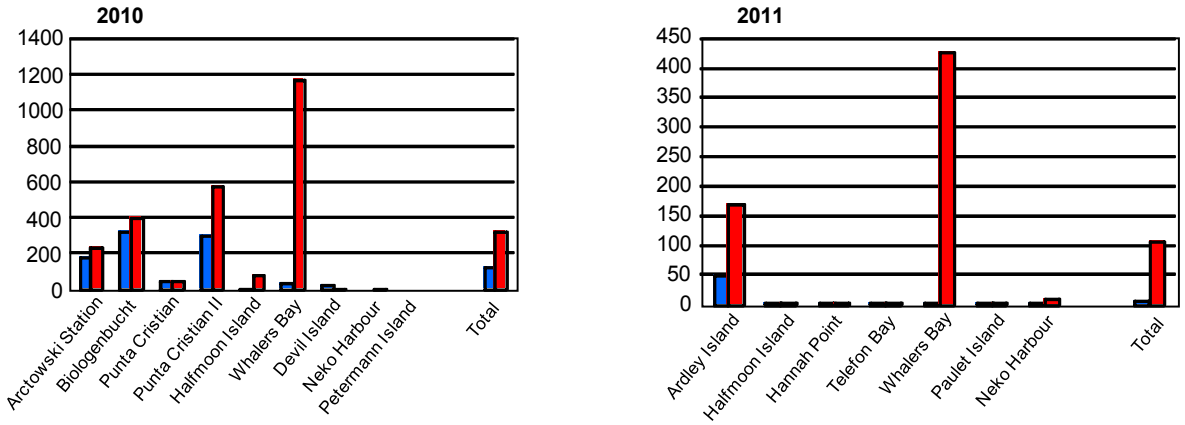


Fig. 37: Total collembolan densities (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Human influenced and non-influenced study areas were most likely also affected by many other factors. Thus, the anthropogenic impact should not only be evaluated directly by simple comparisons of the influenced and non-influenced areas. In this study, covariance analyses (ANCOVAs) were also carried out in order to filter out the effects of these other factors. Since the vegetational cover was one of the most significant factor determining the size and composition of the collembolan communities (as well as those of the other faunal groups), it was always included in the ANCOVAs together with human influence (treatment) as predictive

and dependent factors, respectively. Other factors were considered as covariables. The study years 2010 and 2011 years were analysed separately, since generally different locations were investigated in these two years. As mentioned in the Methods section, due to statistical difficulties with the data, a lack of statistical significance within the covariance analyses does not imply negative results. Thus only positive (statistically significant) results are shown here.

The covariance analysis could also show significant effects of human influence on total collembolan densities in both study years, with higher abundances in the non-influenced areas (Figs 38 and 39; Appendix 6, Table A6-1). This effect of human influence was stronger and negative in plots with medium vegetational cover, as reflected in the significant statistical interaction between vegetational cover and human influence on collembolan densities in both study years (Appendix 6, Table A6-1). In other words, when vegetation cover was only sporadic, humans influenced Collembola more than if no and/or much vegetation was present. These analyses revealed a significant effect of vegetational cover on collembolan densities only in the year 2011 (Appendix 6, Table A6-1), with higher densities in medium levels of vegetational cover. No significant influence of vegetation on densities could be shown in 2010, which - because of the statistical difficulties mentioned above - cannot be interpreted as implying that no effects were present, but rather than simply no judgment can be passed based on these analyses.

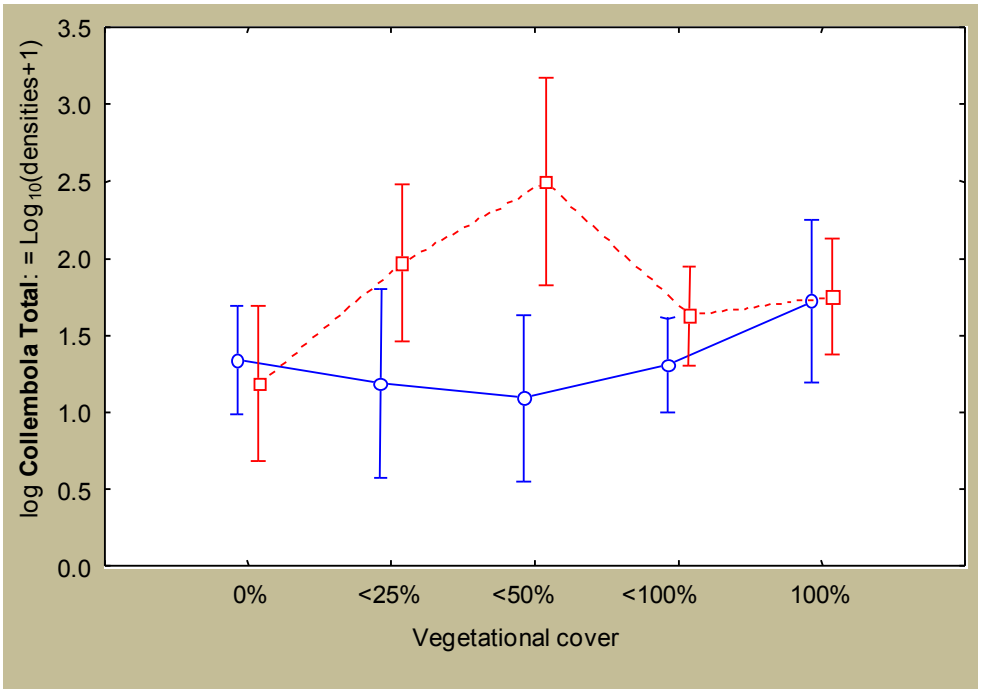


Fig. 38: Results of the covariance analysis (ANCOVA) of total collembolan densities recorded in 2010 after filtering out various background habitat parameters. Densities (given in log individuals per 100 cm^3 substrate) in anthropogenically influenced areas in blue and in non-influenced areas in red. Different letters denote significant collembolan density differences between vegetational cover categories.

A total of 11 collembolan species were identified in the present investigations, 10 in the year 2010 and eight in the year 2011. The collembolan species numbers recorded in the sampled localities are given in Table 7). To minimize the specifity of local sampling, the locations were grouped together into three areas, notated here as: King George Island (5 sampled localities), Livingston Island (3 sampled localities), Deception Island (3 sampled localities), Antarctic

Peninsula (3 sampled localities) and Weddell Sea (2 sampled localities). According to the data of the present project, Petermann Island is devoid of Collembola. Nevertheless, the distributional areas of at least five species (*C. antarcticus, F. grisea, F. octooculata*, and less probably *C. badasa* and *A. brucei*) also cover this location. It can be assumed that the sampling sites were not suitable for the survival of Collembola (i.e., absence of visible ground vegetation). If more favourable biotopes exist on these islands, they may be likely inhabited by Collembola.

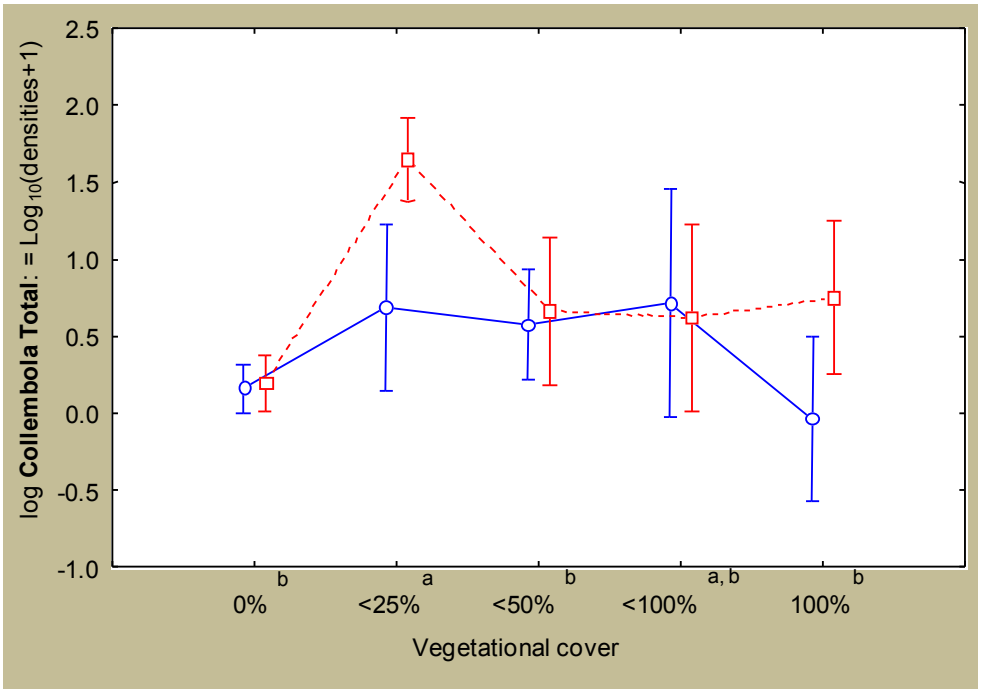


Fig. 39: Results of the covariance analysis (ANCOVA) of total collembolan densities recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Table 7: Number of collembolan species (N) in different sampling locations.

Area	Location	N
King George Island	Arctowski station	6
	Biologenbucht	6
	Punta Cristian	4
	Punta Cristian 3	5
	Ardley Island	5
	all locations together	**6**
Livingston Island complex	Halfmoon Island (2010)	3
	Halfmoon Island (2011)	2
	Hannah Point	3
	all locations together	**5**
Deception Island	Whalers Bay (2010)	6
	Whalers Bay (2011)	7
	Telefon Bay	2
	all locations together	**8(11)***
Antarctic Peninsula in Danco and Graham coast areas	Petermann Island	0
	Neko Harbour (2010)	1
	Neko Harbour (2011)	2
	all locations together	**2**
Weddell Sea	Devil Island	4
	Paulet Island	1

* value including literature data is given in parenthesis

Significant differences were also observed in species richness (as average number of species per sample) among the localities (Fig. 40; Appendix 5, Table A5-1). The highest species richnesses were observed 2010 in the study sites on King George Island (Fildes Peninsula) and 2011 particularly on Ardley Island, Whalers Bay (Deception Island) and Neko Harbour. Although slightly more species were identified in the first study year, significant differences in total identified species number did not exist between the two study years, also not in those sites investigated in both years (Appendix 5, Table A5-1).

As in the total densities, local species diversity, at the level of a single core, also showed a strong negative correlation to latitude (Appendix 4, Table A4-1), indicating greater species richness at lower latitudes (Fig. 36, right). The decline of diversity at higher latitudes can be revealed from the data of the present project (Fig. 41, left), from 6 species on King George Island to 2 on the Danco and Graham coasts. When data is included in this analysis (Fig. 41, right), the influence of latitude and longitude is no longer as distinct. In both approaches (Fig. 41, left and right) the fauna of one area, Deception Island, is obviously the species richest, which appears to be an anomaly but can be explained by, e.g., the special anthropogenic and climatic conditions there (see Discussion of Deception Island below).

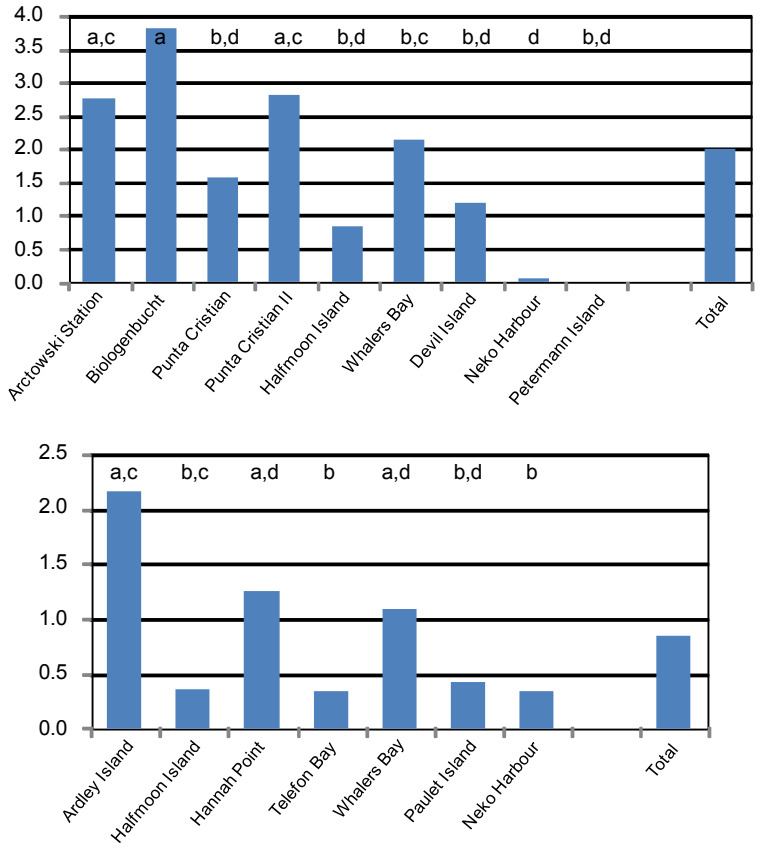


Fig. 40: Total collembolan species richness (in average species per area) observed in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

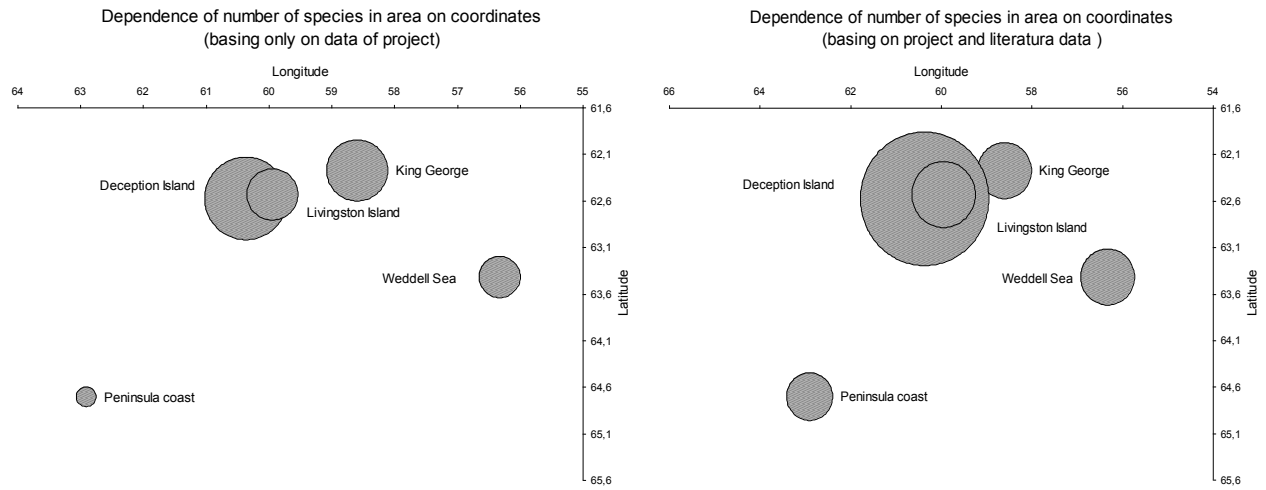


Fig. 41: Number of collembolan species in different sampling areas according to data of the present project (left) and including literature data (right). The diameter of a circle is proportional to species richness.

Vegetational cover interacted as strongly with species richness as it did with total densities, with large correlation coefficients in both study years (0.521 in 2010 and 0.479 in 2011; Appendix 4, Table A4-1), indicating that species-richer collembolan communities occurred under denser vegetational cover. The influence of vegetational cover was most obvious for indigenous species, while non-indigenous showed an opposite pattern (Fig. 42; see results of individual species below for a classification of indigenous and non-indigenous species). This shows that indigenous species were more abundant on plots with greater vegetational cover, while non-indigenous species preferred barren soil surfaces. According to the positive correlation coefficients, vegetational cover presumably determined the level of species richness and abundance of native (indigenous) populations of Collembola. The species diversity of the vegetation was apparently a less significant factor (Fig. 43): on bare ground (no plant species) communities of 2-3 species of Collembola often occurred, while even under only one plant species collembolan communities of 2-5 species could develop. Total species richness also correlated negatively with coarser grained soil substrates and positively with finer grain sizes, but only in 2011 (Appendix 4, Table A4-1). Soil moisture obviously also positively influenced total collembolan species richness in both years. Species richness reacted positively to both quantities and qualities of soil organic matter with high and significant correlation coefficients particularly in 2010: the average number of Collembola species was higher in nutrient-richer soils. The reaction of particular species depended on the species group. Indigenous species mostly followed the general trend of a positive reaction to soil nutrient status, while non-indigenous species, *H. viatica* and *C. caecus*, correlated negatively (see below). Total species richness of the collembolan communities were also apparently affected negatively by pH in 2010.

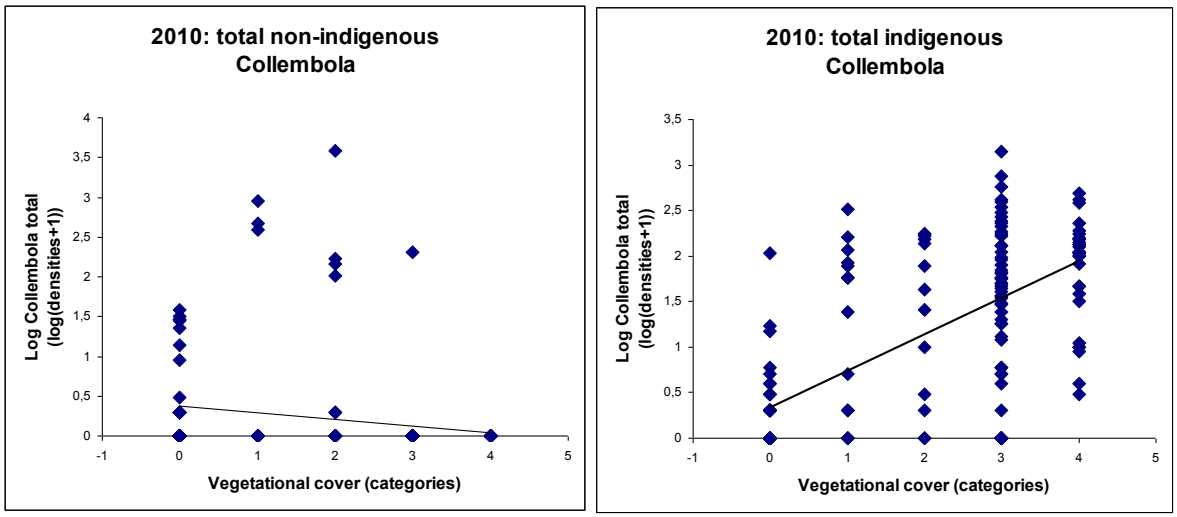


Fig. 42: Correlation between total density of indigenous (left) and non-indigenous (right) species and vegetation cover in 2010. For the definition of indigenous and non-indigenous species see below (Results of individual species) and Table 8.

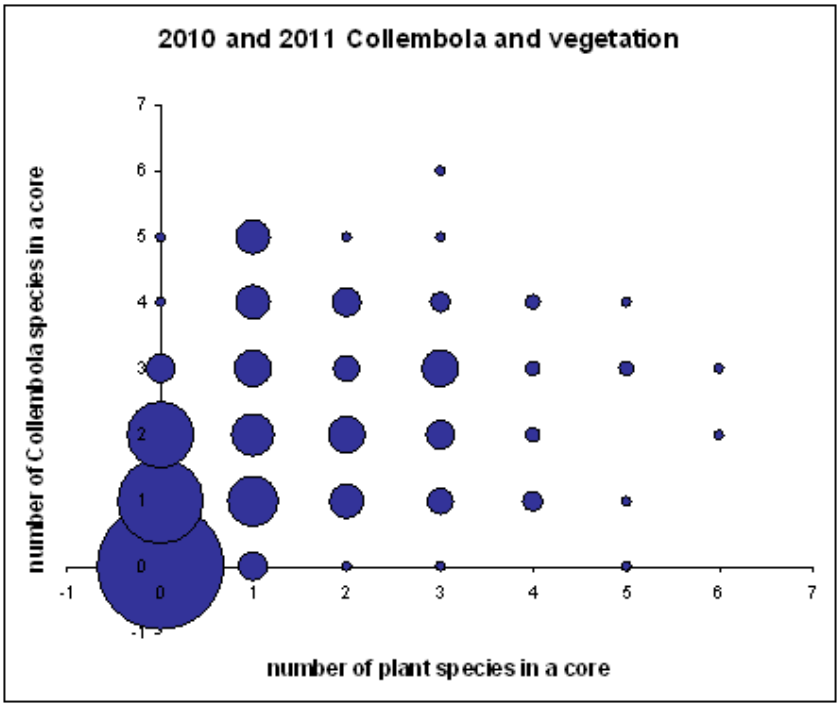


Fig. 43: Species richness of Collembola and plants (square of a circle is proportional with the frequency of the occurrence of the combination of number of species of Collembola with species of plants).

Anthropogenic influence did not always affect total species richness. Only in the year 2011 were overall significantly higher species numbers in areas *not* influenced by humans observed (Fig. 44; Appendix 5, Table A5-3). In this year, the human influence was apparently strongest at Whalers Bay (Deception Island).

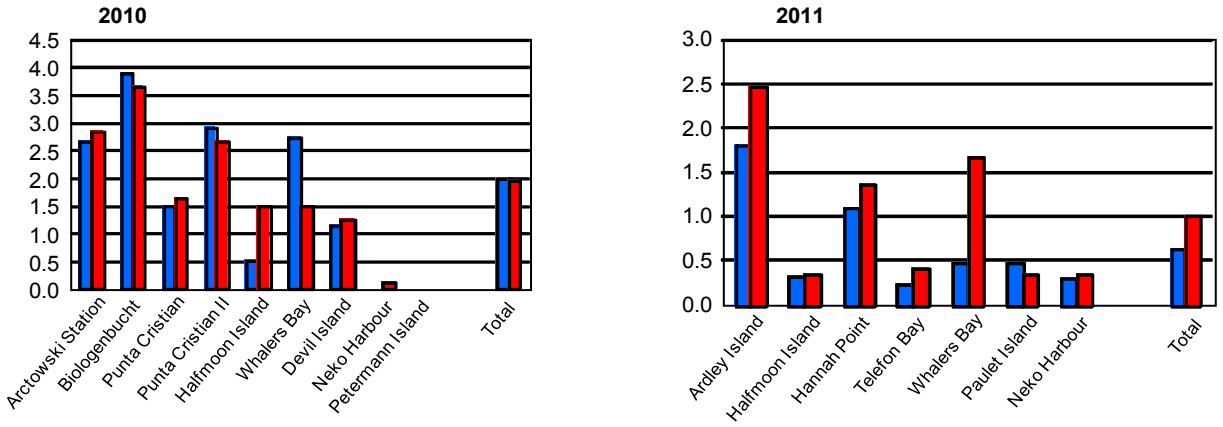


Fig. 44: Total collembolan species richness (in average species number per area) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

3.4.1.2 Results and descriptions of the determined species

The collembolan species recorded in the present project are shown in their systematic position in the following. Systematics follow Janssens (2012). Their average densities in the various localities are given in Appendix 3, Table A3-1.

<u>Poduromorpha</u>

Hypogastruridae

Hypogastrura viatica (Tullberg, 1872)

Neanuridae

Frieseinae

Friesea grisea (Schäffer, 1891)
Friesea woyciechowskii Weiner, 1980

Tullbergiidae

Tullbergiinae

Tullbergia mixta Wahlgren, 1906

Stenaphorurinae

Mesaphorura macrochaeta Rusek, 1976

Onychiuridae

Onychiurinae

Deuteraphorura cebennaria (Gisin, 1956) *

<u>Entomobryomorpha</u>

Isotomidae

Anurophorinae

Cryptopygus antarcticus Willem, 1901
Cryptopygus badasa Greenslade, 1995
Cryptopygus caecus Wahlgren, 1906

Proisotominae

Archisotoma brucei (Carpenter, 1907)
Proisotoma minuta (Tullberg, 1871)

Isotominae

Folsomotoma octooculata (Willem, 1901)

* species determined during the present project, but sampled separately by the British Antarctic Survey.

The determined species can be grouped as follows: indigenous Antarctic species widely distributed in the Antarctic or/and Subantarctic, indigenous Antarctic species more locally distributed (local endemics), and non-indigenous species (Table 8).

Regarding the first mentioned group (native species), it must be remarked that, in fact, only two species have been previously known to be distributed throughout the continental Antarctic: *Cryptopygus antarcticus* and *Friesea grisea*. The records of the latter in Eastern Antarctic have been practically refuted based on their morphology (Deharveng, 1981 and subsequent publications), so that this species most likely does not occur in the Eastern Antarctic. The same has been fairly well proven by Torricelli et. al. (2010) for *Friesea grisea,* the "only species which has been described for both major regions of the continent". Thus, up to now, probably no species of Collembola occurs in both western (the maritime) and eastern parts of Antarctica.

In the following sections, the individual collembolan species recorded in the present project are described together with information on their known distribution. As opposed to the presentation of the Acari (mite) species recorded in the present project (see chapter 3.4.2), due to the relatively similar ecological preferences of these species, these aspects are given in the discussion (s. Chapter 4). The species are grouped in the following according to their distribution as given in Table 8, with species new to and potentially non-native in the Antarctic described in the last group of this section.

Table 8: Indigenous and non-indigenous Collembola occurring in the maritime Antarctic

Indigenous species	widely distributed	*Cryptopygus antarcticus* Willem, 1901 *Friesea grisea* (Schäffer, 1891) *Archisotoma brucei* (Carpenter, 1907)
	locally distributed	*Cryptopygus badasa* Greenslade, 1995 *Tullbergia mixta* Wahlgren, 1906 *Friesea woyciechowskii* Weiner, 1980 *Folsomotoma octooculata* (Willem, 1901)
Non-indigenous species	high risk status	*Hypogastrura viatica* (Tullberg, 1872)
	middle risk status	*Protaphorura fimata* * (Gisin, 1952) *Folsomia candida* * Willem, 1902 *Mesaphorura macrochaeta* ** Rusek, 1976 *Proisotoma minuta* ** (Tullberg, 1871) *Deuteraphorura cebennaria* ** (Gisin, 1956)
group and status unclear		*Cryptopygus caecus* *** Wahlgren, 1906

* species recorded from the maritime Antarctic in the literature and not identified in the present project

** species recorded for the first time in the maritime Antarctic during the present project

*** placed in intermediate position between indigenous and non-indigenous species (see the remarks to the individual species)

3.4.1.2.1 Indigenous Antarctic species widely distributed in the Antarctic and Subantarctic

Cryptopygus antarcticus Willem, 1901

C. antarcticus was recorded in eight localities of the present project. It belongs to the nominative subspecies *antarcticus antarcticus*. It was described from a range of localities in the Gerlache Strait, all of which are about 200 km south of the South Shetland Islands (Augustus

Island, Harry Island, Danco Territory, Brabant Island, Cap van Beneden, Ile de Cavelier de Cuverville, Wiencke Island and Bob Island). No type locality was designated. The species is widely distributed in the Maritime Antarctic, commonly occurring in the South Shetland Islands. It was previously incorrectly recorded from the Eastern Antarctic. In a broad understanding, it was recorded throughout the Subantarctic, but the identity of these populations is doubtful due to molecular and morphological differences found in modern studies (Stevens et al., 2005; Greenslade, 2006, Deharveng, 1981). In Eastern Antarctica other species of the genus (*C. cisantarcticus, C. sverdrupi*) or other collembolan taxa replace *C. antarcticus*. Records of this species' distributions are shown in Figs 45 and 46. Based on modern knowledge, the distribution of this species ranges from Subantarctica to the Antarctic Peninsula. It was recorded in Hannah Point and Neko Harbour – und thus further south - for the first time in the present project.

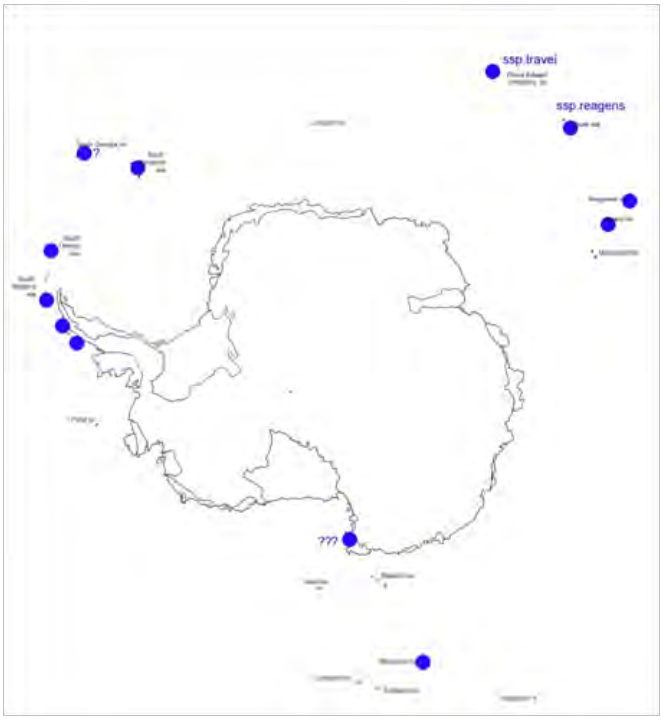


Fig. 45 (left): Records of *C. antarcticus* along the Antarctic Peninsula. Areas within the possible type locality marked by dotted line.

Fig. 46 (right): Records of *C. antarcticus* throughout the entire Antarctic and Subantarctic. "???" = uncertain records.

The occurrence of *C. antarcticus* correlated particularly (positively) with amounts of organic matter in soils, soil moisture as well as finer grained soil substrates (sands) in both study years (Appendix 4, Table A4-1). In the year 2010 the species also correlated with latitude (= locality),

while its records from the year 2011 also related positively to vegetation cover and negatively to pH.

This species was generally found in higher abundances in areas not influenced by humans. However, this was statistically significant only in the year 2011 or when both years were analyzed together (Fig. 47; Appendix 5, Table A5-4).

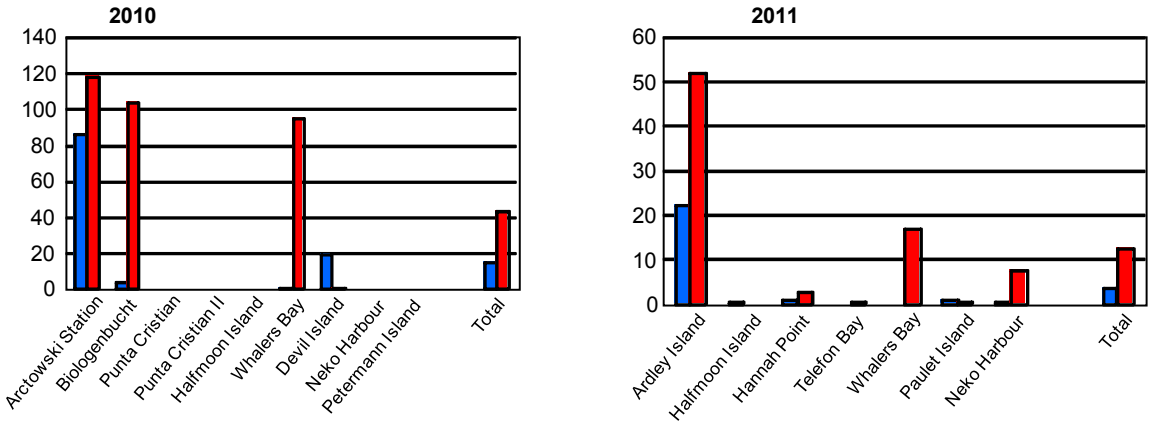


Fig. 47: Total densities of *C. antarcticus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

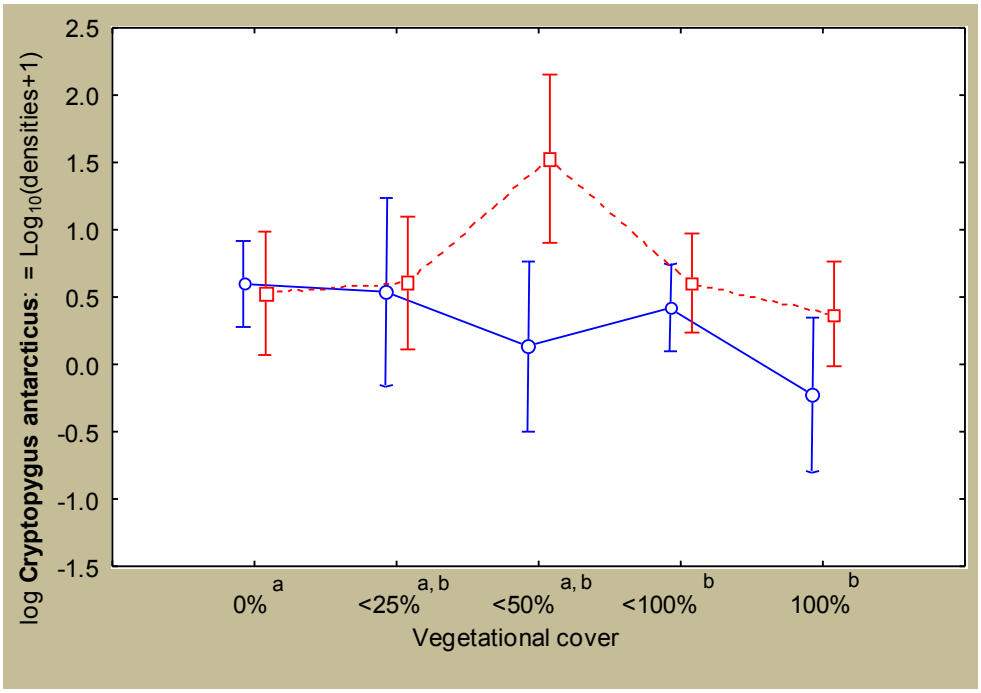


Fig. 48: Results of the covariance analysis (ANCOVA) of the densities of *C. antarcticus* recorded in 2010. Figure explanation as in Fig. 38.

Filtering out various background habitat parameters, the covariance analyses revealed results similar to those of the pure variance analyses. For instance, significantly higher densities in anthropogenically non-influenced areas were determined in the year 2010 (Fig. 48; Appendix 6, Table A6-1). In the year 2011, a significant relationship between the densities of *C. antarcticus* and vegetational cover could be determined (Fig. 49; Appendix 6, Table A6-1), with significantly higher densities at medium levels of vegetational cover. In both years a significant

interaction between human influence and vegetational cover could be revealed, whereby significantly higher densities in non-influenced areas occurred in sites with low to medium levels of vegetational cover.

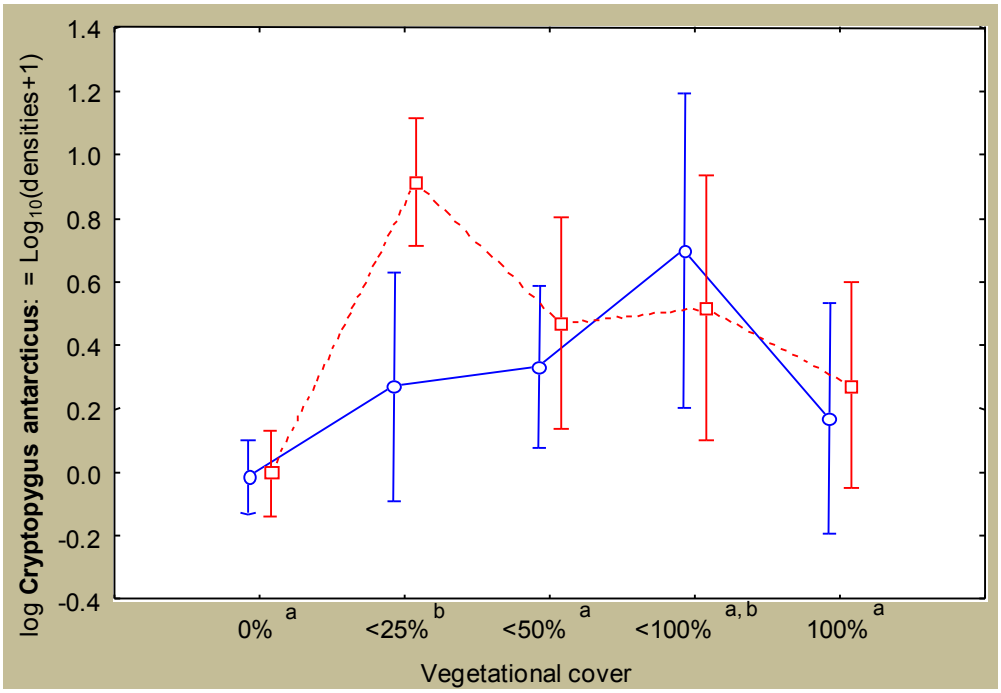
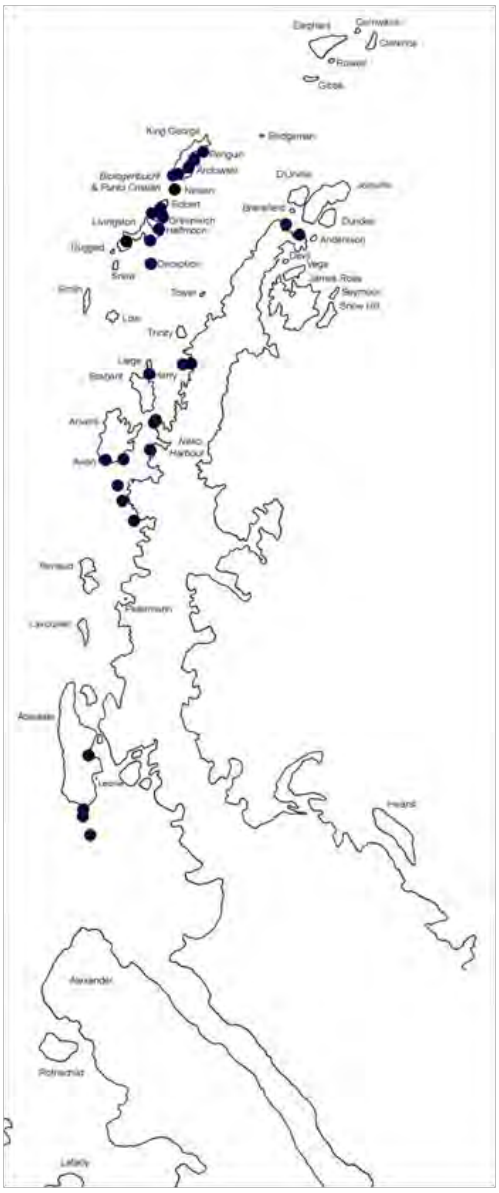


Fig. 49: Results of the covariance analysis (ANCOVA) of the densities of *C. antarcticus* recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Friesea grisea (Schäffer, 1891)

This species was recorded in eight locations in the present investigations. It was first described as a new species from South Georgia, although the description was incomplete. Many records and redescriptions have been made from other locations (for details see Greenslade, 2010). Thus far, the species has not been recollected from South Georgia recently, despite several who surveys. The true identity of this species in the Maritime Antarctic remains doubtful. A possible alternative name for these records of *Friesea grisea* is *Friesea antarctica* (Willem, 1901), the latter having been described from more southern areas as *Achorutoides antarcticus*. The species is widely distributed in the maritime Antarctic and is common in the South Shetland Islands. Up to now it has been the only "pan-Antarctic" species of Collembola recorded both from Western and Eastern Antarctic. However, large molecular differences were shown between western and eastern populations, which strongly restricts the true distribution of *F.grisea* (Torricelli et. al, 2010). Based on modern knowledge, this species will possibly receive the status of a Western Antarctic species. Unlike *C. antarcticus*, the species was not recorded in the Subantarctic (with the exception of South Georgia). Records of its distribution are shown in Figs 50 and 51.



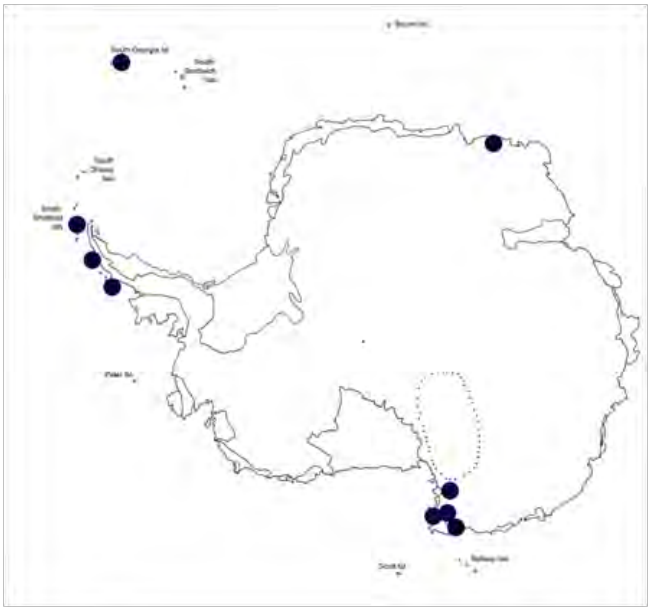


Fig. 50 (left): Records of *F. grisea* in the maritime Antarctic.
Fig. 51 (right): Records of *F.grisea*. The area on Victoria Land where the species is absent with certainty marked with a dotted line.

F. grisea correlated in both study years to latitude (= locality) as well as positively to vegetational cover and soil moisture and negatively to coarser grained substrates (gravels) (Appendix 4, Table A4-1). In 2010 the species further correlated positively to various parameters concerning the quantity and quality (C/N-ratio) of organic material in the sampled soils, while in 2011 it also correlated positively to finer grained substrates (sands) as well as negatively to soil pH.

The human influence on the abundances of this species was dependent on locality. In some localities it occurred in higher densities in human influenced sites, while in other localities its highest densities were found in the non-influenced areas. Therefore, overall, no statistically significant influence of human activity could be determined by this analysis (Fig. 52; Appendix 5, Table A5-4).

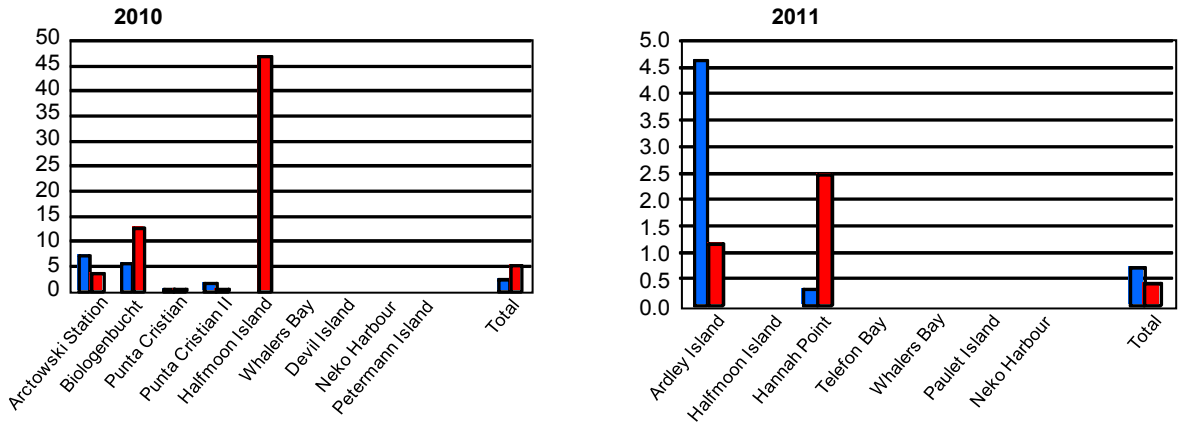


Fig. 52: Total densities of *F. grisea* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

In contrast to the variance analyses, the covariance analyses could reveal a statistically significant effect of human influence on the overall densities of *F. grisea* in both years (Figs. 53 and 54; Appendix 6, Table A6-1), with higher densities in the anthropogenically non-influenced areas. In 2011 this species' densities were furthermore significantly larger at medium-high levels of vegetational cover (Appendix 6, Table A6-1). Also in this year, a significant interaction between *F. grisea*'s densities and vegetational cover could be determined (Appendix 6, Table A6-1), whereby the significantly higher densities in the non-influenced areas were particularly within areas with medium-high vegetational cover.

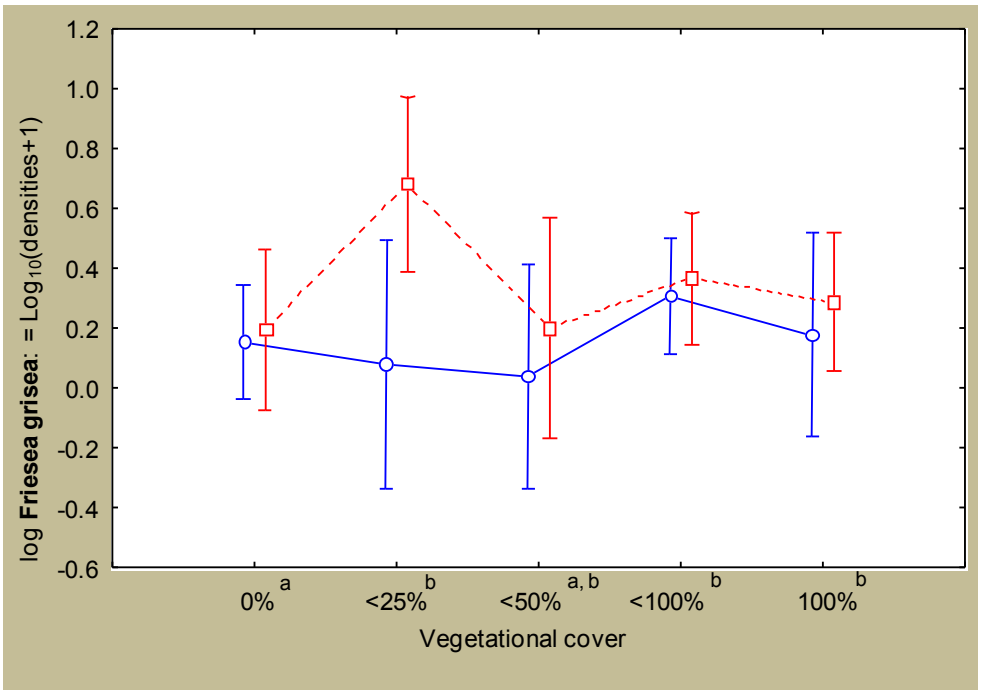


Fig. 53: Results of the covariance analysis (ANCOVA) of the densities of *F. grisea* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

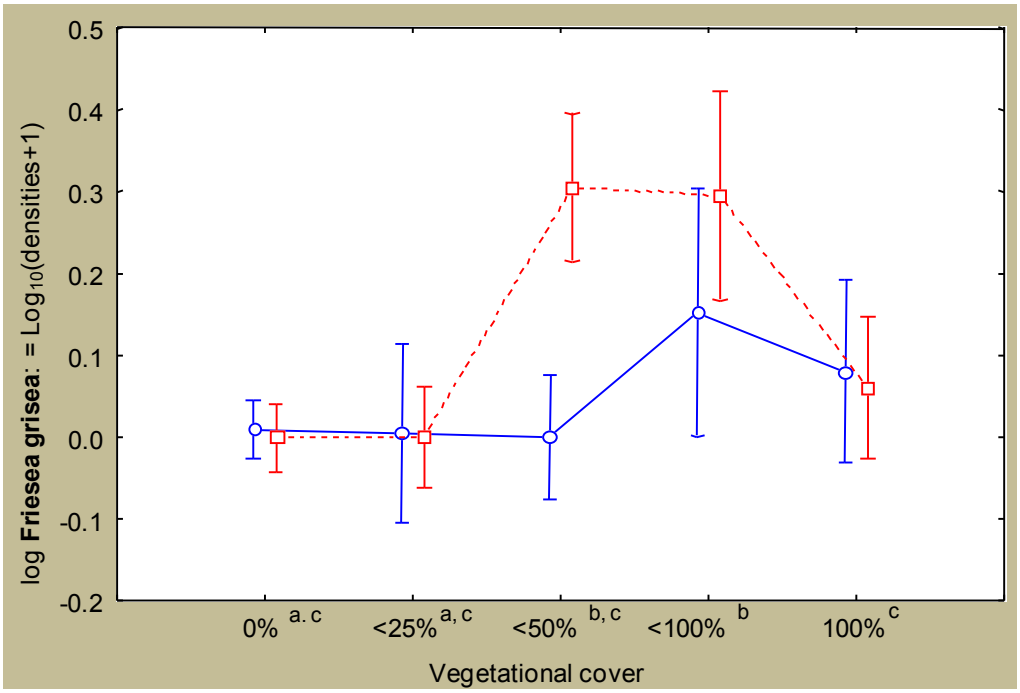


Fig. 54: Results of the covariance analysis (ANCOVA) of the densities of *F. grisea* recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Archisotoma brucei (Carpenter, 1907)

A. brucei was recorded in two locations during this study. Its distribution is similar to that of *C. caecus*, as it is widely known from the Subantarctic but less so from the continental Antarctic; it has been recorded in New Zealand. It was first described from Laurie Island (South Orkney Islands). It is a littoral species, which probably explains why it was scarce in the present samples. Records of its distribution are shown in Figs 55 and 56. The species was recorded in Devil Island for the first time in the present study.

The occurrence of *A. brucei* only correlated to habitat parameters in the year 2010 (Appendix 4, Table A4-1). In this year the species correlated with parameters concerning locality (i.e., sampling date, longitude). In contrast to most other collembolan species, *A. brucei* correlated in this year positively to pH and negatively to finer grained soil textures (i.e., positively to silt and clay and negatively to medium grained sands).

Due to the fact that this species was only recorded in few individuals, no statistical analysis of a potential anthropogenic influence on its abundances could be undertaken.

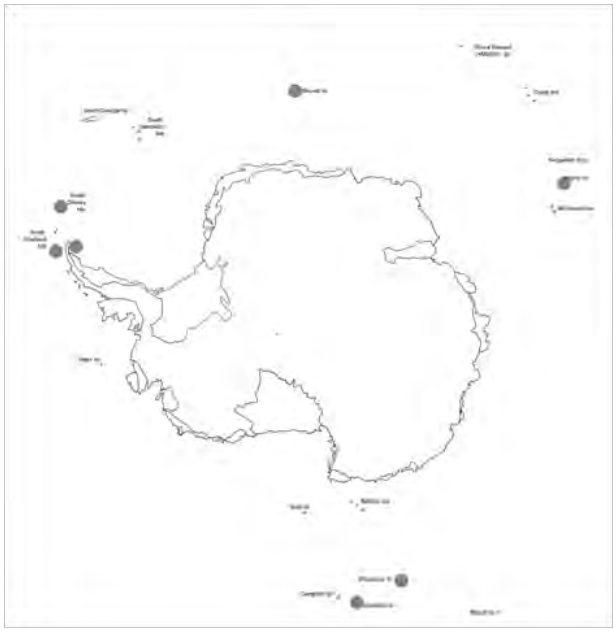


Fig. 55 (left): Records of *A. brucei* in the maritime Antarctic.
Fig. 56 (right): Records of *A. brucei* throughout the entire Antarctic and Subantarctic

3.4.1.2.2 More locally distributed indigenous Antarctic species

Cryptopygus badasa Greenslade, 1995

This species was recorded in eight locations in the present study. The species was first described from Livingston Island (South Shetland Islands). Later it was scarcely recorded throughout the western part of the maritime Antarctic (not recorded in South Orkney Islands, South Sandwich Islands or Bouvet Island). It is a local species for part of the maritime Antarctic. It can be assumed that some older records of *C. antarcticus* in the maritime Antarctic refer to this species. *C. badasa* shares many morphological characters with the latter, but readily differs in smaller size, paler colouration and slender body. Records of its distribution are given in Figs 57 and 58. The species was recorded in Devil Island and Hannah Point for the first time during the present investigation.

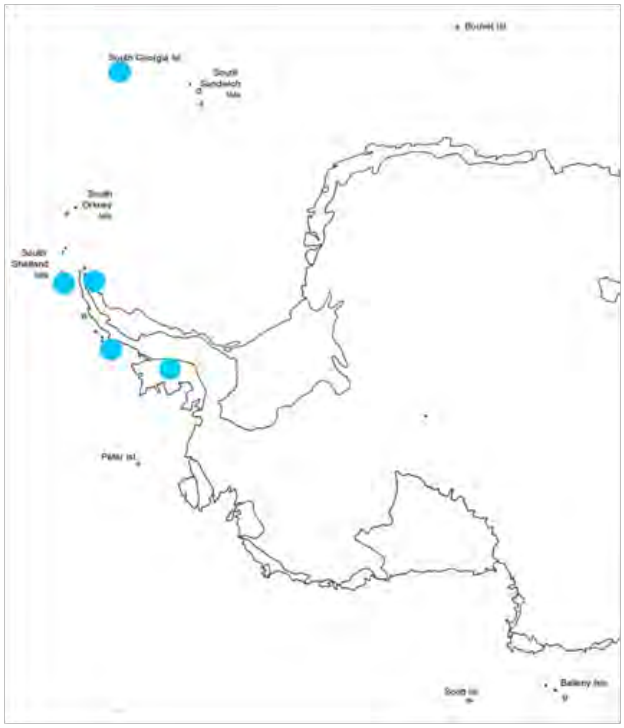


Fig. 57 (left): Records of *C. badasa* in the maritime Antarctic.
Fig. 58 (right): Records of *C. badasa* in the Antarctic and Subantarctic.

C. badasa correlated strongly in both years to factors regarding location (i.e., sampling date, latitude) as well as positively to vegetational cover (Appendix 4, Table A4-1). In the year 2010 it further correlated positively to factors regarding the content and quality of organic material in the sampled soils as well as negatively to temperature. In the year 2011 it also correlated positively to soil moistures.

Although the species was often found in higher individual numbers in anthropogenically non-influenced areas, due to high sample-to-sample variability a human influence on its densities was statistically not significant according to this analysis (Fig. 59; Appendix 5, Table A5-4).

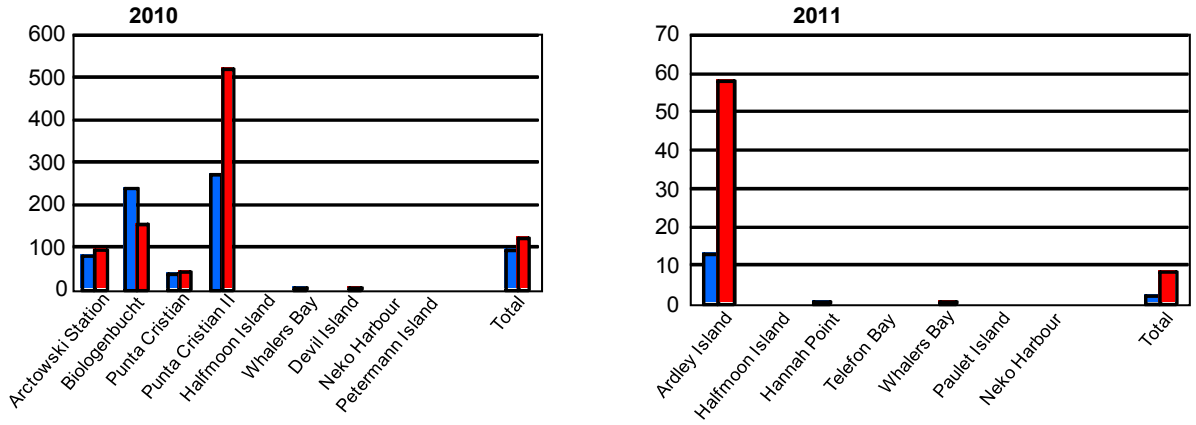


Fig. 59: Total densities of *C. badasa* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analyses, on the other hand, revealed somewhat contradictory results to the pure variance analyses. Firstly, filtering out various habitat parameters revealed significantly higher densities of *C. badasa* at medium and high levels of vegetational cover (Figs 60 and 61; Appendix 6, Table A6-1). In 2011, furthermore, significantly higher densities of *C. badasa* were actually found in anthropogenically influenced areas, whereby these mostly occurred at medium-high levels of vegetational cover (interaction between human influence and vegetational cover.

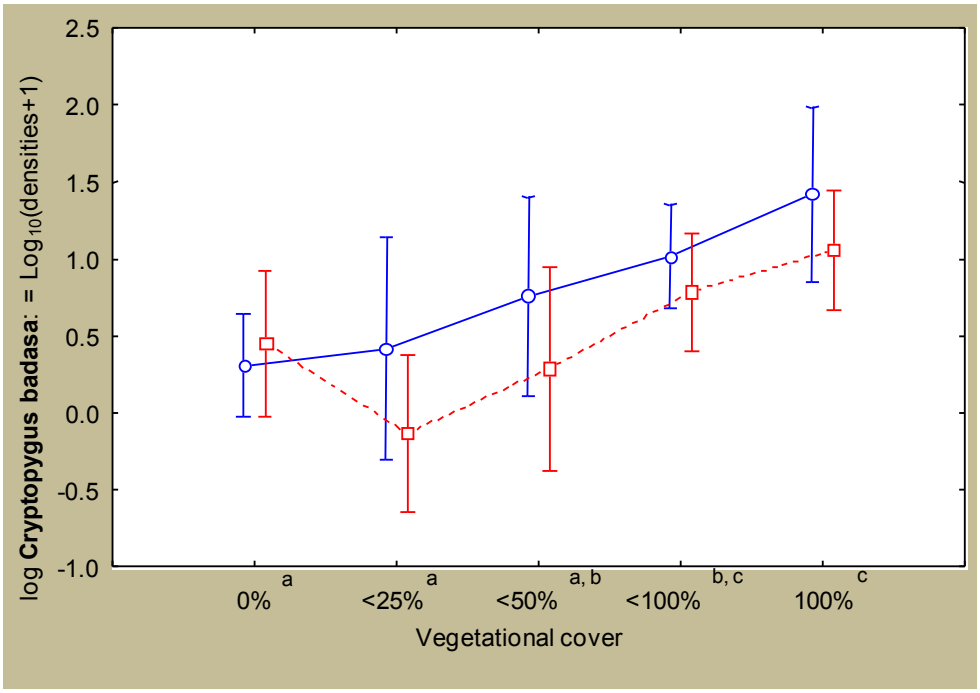


Fig. 60: Results of the covariance analysis (ANCOVA) of the densities of *C. badasa* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

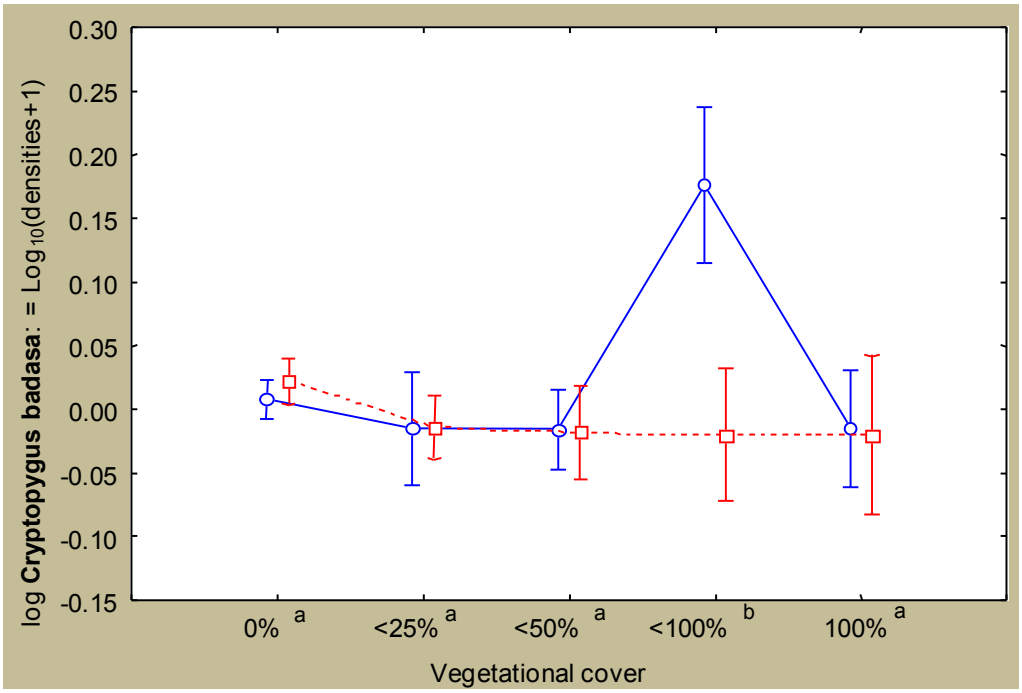


Fig. 61: Results of the covariance analysis (ANCOVA) of the densities of *C. badasa* recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Tullbergia mixta Wahlgren, 1906

This species was recorded in six locations in the present study, five of which are on King George Island. It was first described from Nelson Island (South Shetland Islands). In the opinion of Greenslade (2010), it was probably misidentified as *T. mediantarctica* in material from King George Island. It is most likely endemic to the South Shetland Islands. For all of its known records, see Fig. 62.

T. mixta correlated to factors concerning locality as well as positively to vegetational cover in both years (Appendix 4, Table A4-1). In the year 2010 in further correlated positively to amounts and quality of organic material and soils and 2011 positively to soil moistures. The correlation results regarding soil temperature, however, were contradictory, with a negative correlation in the year 2010 and a positive correlation 2011.

This species was usually found in higher individual numbers in anthropogenically non-influenced areas. A human influence on its densities was statistically significant only in the year 2011 or when both years were sampled together (Fig. 63; Appendix 5, Table A5-4). The covariance analyses could primarily show significantly higher densities of *T. mixta* at high levels of vegetational cover in the year 2010 (Fig. 64; Appendix 6, Table A6-1).

Fig. 62: Records of *T. mixta* in the maritime Antarctic.

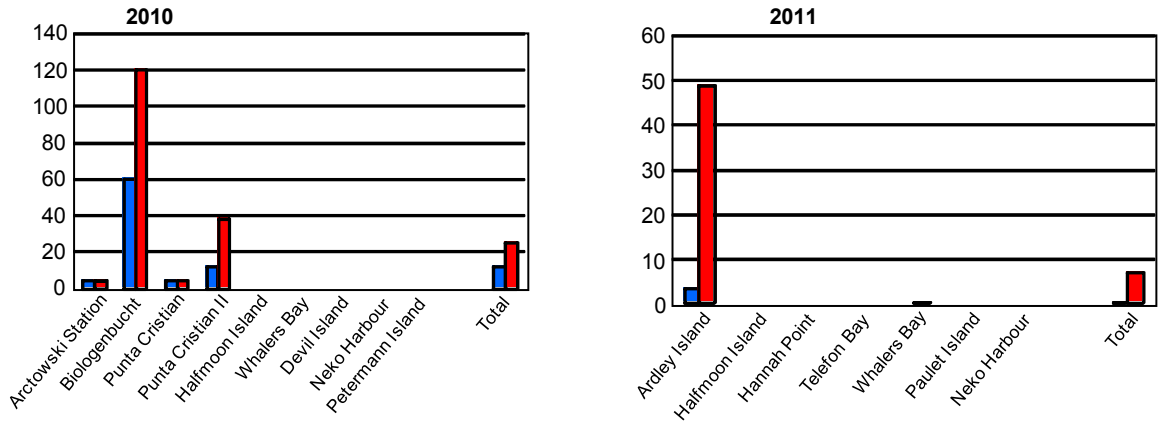


Fig. 63: Total densities of *T. mixta* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

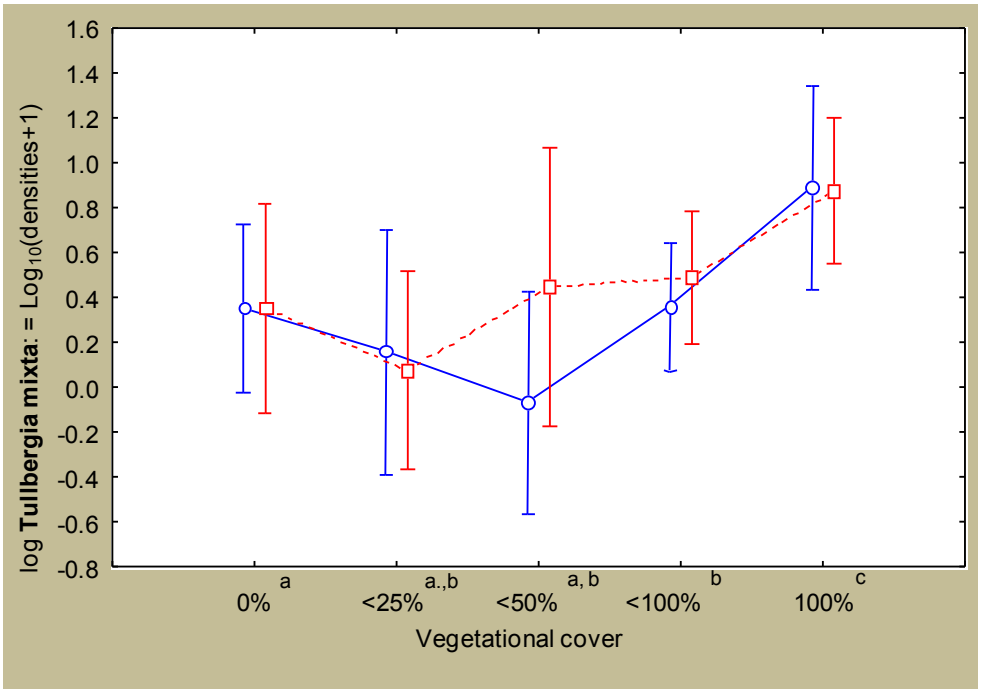


Fig. 64: Results of the covariance analysis (ANCOVA) of the densities of *T. mixta* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Friesea woyciechowskii Weiner, 1980

The species has only been recorded a few times (three in the present study) and is probably endemic to the warmer parts of the Maritime Antarctic (South Shetland and South Orkney Islands). It was first described from King George Island. For all of its known records, see Figs 65 and 66.

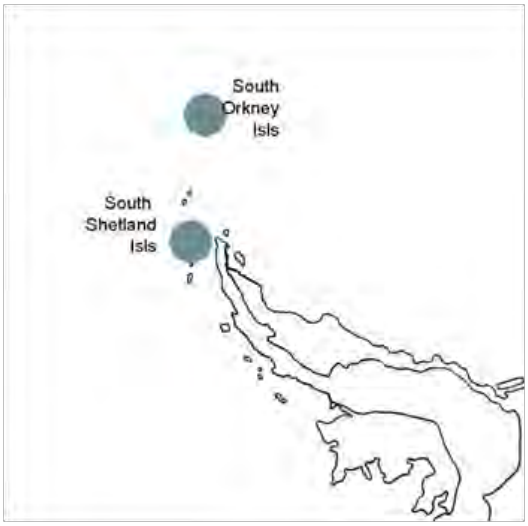


Fig. 65 (left): Records of *F. woyciechowskii* in the South Shetland Islands.
Fig. 66 (right): Records of *F. woyciechowskii* in the maritime Antarctic.

F. woyciechowskii was only recorded in 2010. In this year it only correlated with sampling date (Appendix 4, Table A4-1), which most likely reflects location. It was always found in higher total abundances in anthropogenically non-influenced areas, but due to high sample-two-sample variability this was statistically not significant (Fig. 67; Appendix 5, Table A5-4).

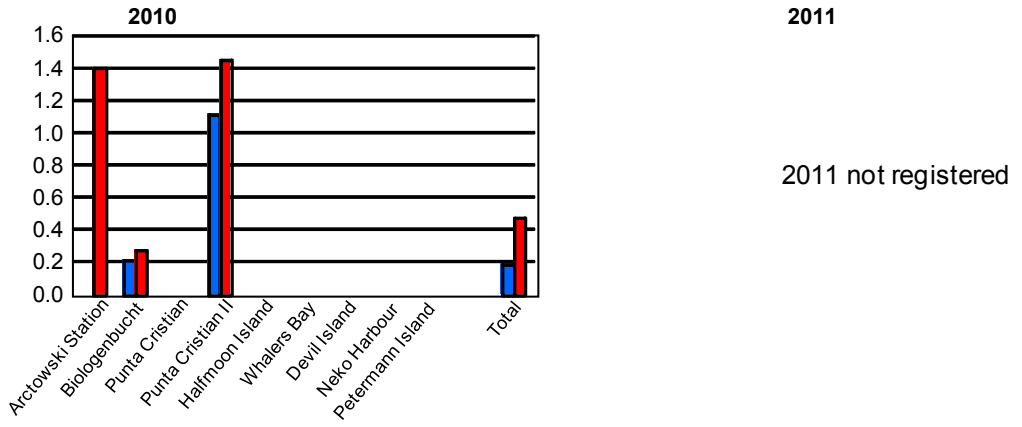


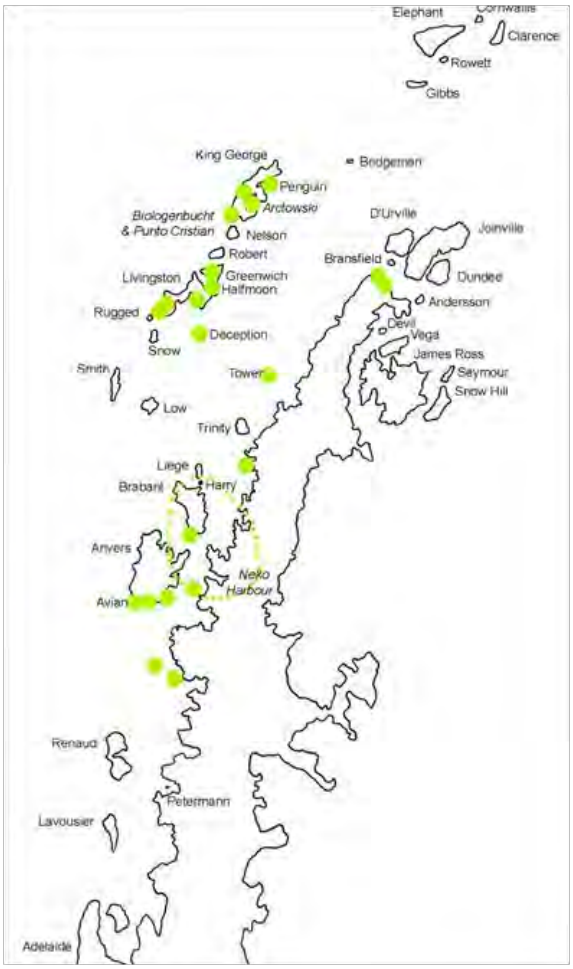
Fig. 67: Total densities of *F. woyciechowskii* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 (no individuals registered in 2011).

Folsomotoma octooculata (Willem, 1901)

F. octooculata was recorded in six locations in this study. It was described as *Isotoma octooculata* by Willem (1901) from Harry Island, Cape van Benenden in Danco Land and Cavelier de Cuverville Island (all in Gerlache Strai). It has only been recorded from the Maritime Antarctic, including the South Shetland Islands and South Orkney Islands, and can be considered a local species. For all of its records see Figs 68 and 69.

This species also correlated to latitude as well as positively to vegetational cover in both study years (Appendix 4, Table A4-1). It showed a positive relation to amounts of organic material and a negative relation to finer grained soil substrates (sands) in 2010 as well as a positive relation to soil moisture in 2011. The correlations to soil temperature were contradictory, with a negative correlation coefficient in 2010 and a positive coefficient in the year 2011.

F. octooculata was often found in higher abundances in the areas not influenced by humans (Fig. 70). However, this was not always the case, as in some localities the species was recorded in higher densities in the anthropogenic influenced areas. Due to these locality-specific differences in its distribution among the anthropogenically influenced and not-influenced areas, no statistically significant human influence on its densities could be determined (2010 $Xr^2 = 0.058$, $P = 0.810$; 2011: $Xr^2 = 0.641$, $P = 0.423$; overall: $Xr^2 = 0.352$, $P = 0.553$).



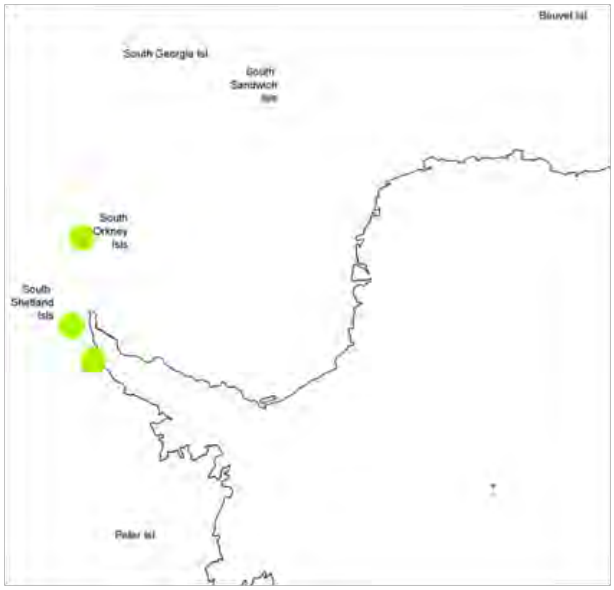


Fig. 68 (left): Records of *F. octooculata* in the maritime Antarctic. Area within the possible type locality is marked by a dotted line.

Fig. 69 (right): Records of *F. octooculata* in the Antarctic and Subantarctic.

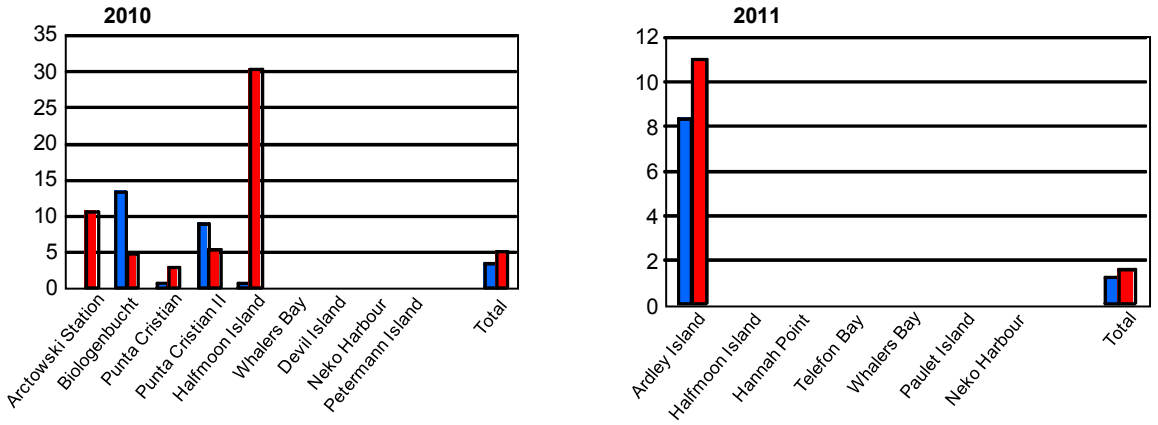


Fig. 70: Total densities of *F. octooculata* (in individuals 100 cm^{-3}) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Through the covariance analysis it could be determined that, overall among all localities, *F. octooculata* occurred in the year 2011 in significantly higher densities when vegetational cover was practically 100% ($F = 7.191$, $P < 0.001$; Fig. 71)

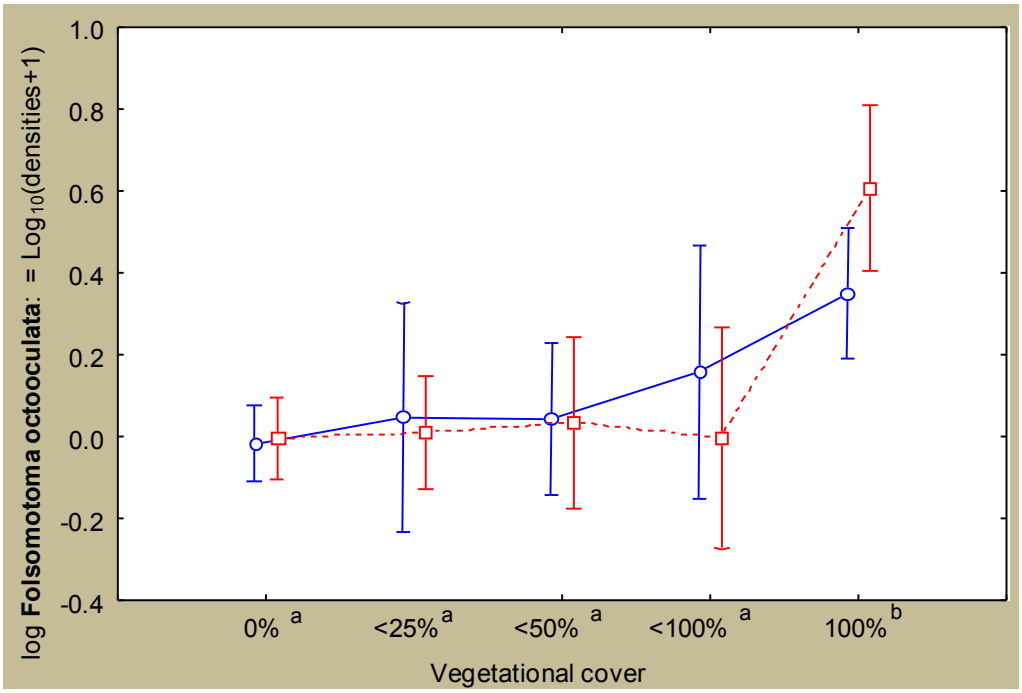


Fig. 71: Results of the covariance analysis (ANCOVA) of the densities of *F. octooculata* recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

3.4.1.2.3 Potentially introduced (non-native) species

Hypogastrura viatica (Tullberg, 1872)

H. viatica is a cosmopolitan species first described from Sweden. It was recorded in this study in four localities (Fig. 72). Sampling during 2011 confirmed the records from 2010 of this species in locations sampled in both study years: Whalers Bay, Neko Harbour and Halfmoon Island. The expansive abilities of the species are described in detail below. For its records in the Antarctic and Subantarctic see Fig. 73.

H. viatica only showed significant correlations to habitat factors in the year 2010 (Appendix 4, Table A4-1). The negative correlations to sampling date and latitude are simply a reflection of locality. In contrast to almost all other collembolan species, the species interestingly showed a positive correlation to soil temperature as well as negative correlations to quantities and qualities of soil organic material. However, this probably does not represent its true habitat preferences, but may simply reflect the conditions on Deception Island, where the species is particularly abundant.

Overall, this species was found in significantly higher densities in anthropogenically non-influenced areas (Fig. 74; Appendix 5, Table A5-4). However, this result was strongly influenced by its very high densities at Whalers Bay (Deception Island), where was particularly abundant in the areas non-influenced by humans.

Filtering out the background habitat parameters in the covariance analyses could reveal that *H. viatica* occurred in significantly higher densities at medium to high levels of vegetational cover in both years (Figs 75 and 76; Appendix 6, Table A6-1).

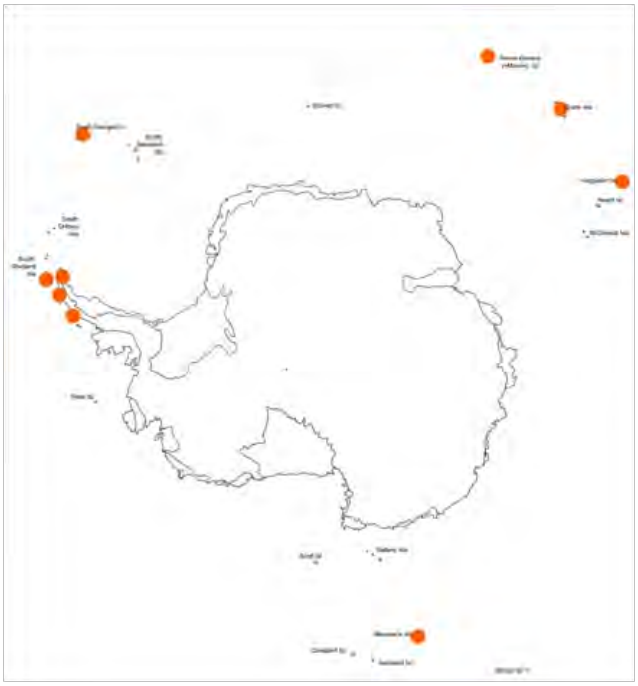


Fig. 72 (left): Records of *H. viatica* in the maritime Antarctic.
Fig. 73 (right): Records of *H. viatica* throughout the Antarctic and Subantarctic.

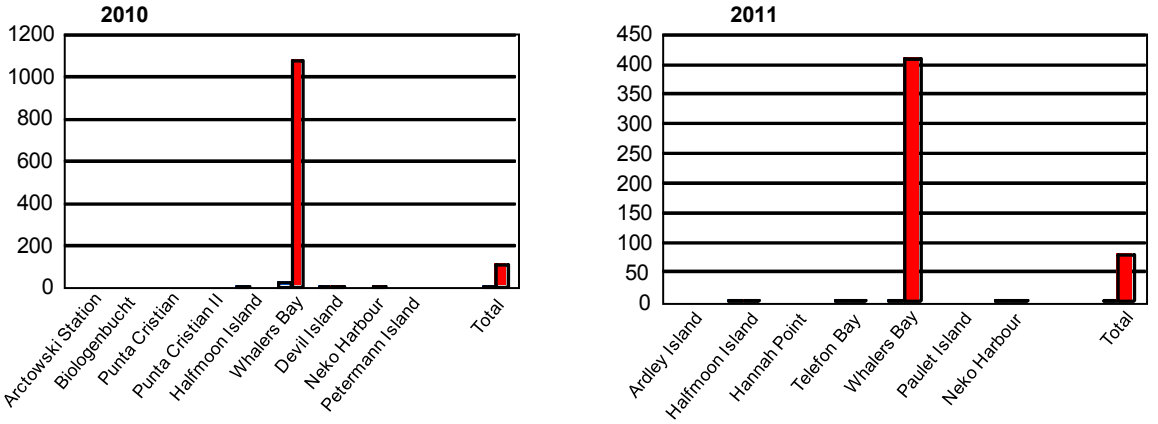


Fig. 74: Total densities of *H. viatica* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

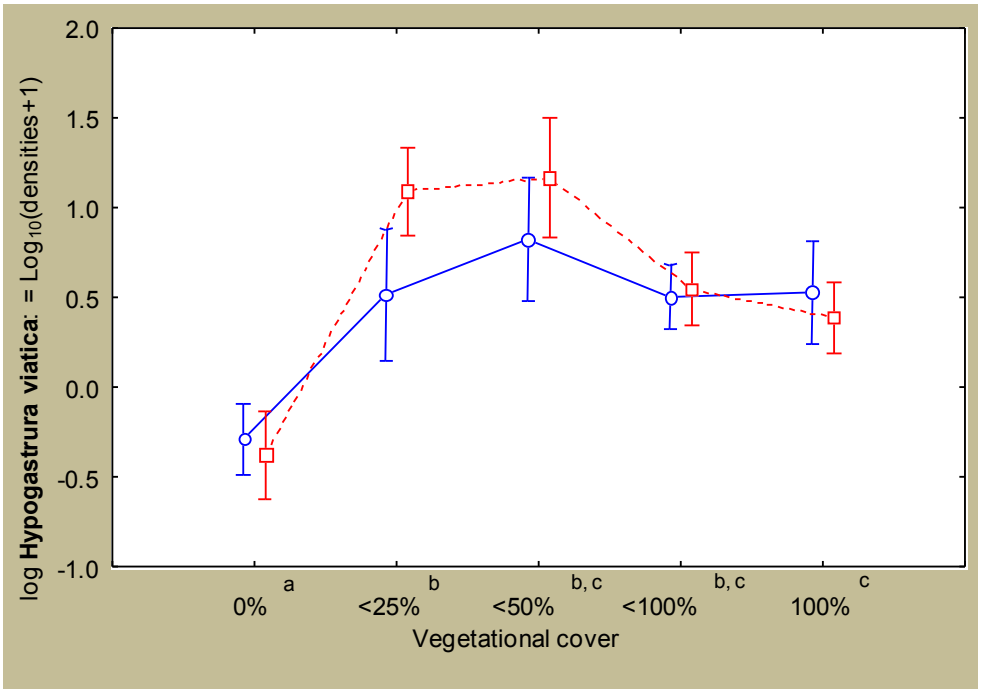


Fig. 75: Results of the covariance analysis (ANCOVA) of the densities of *H. viatica* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

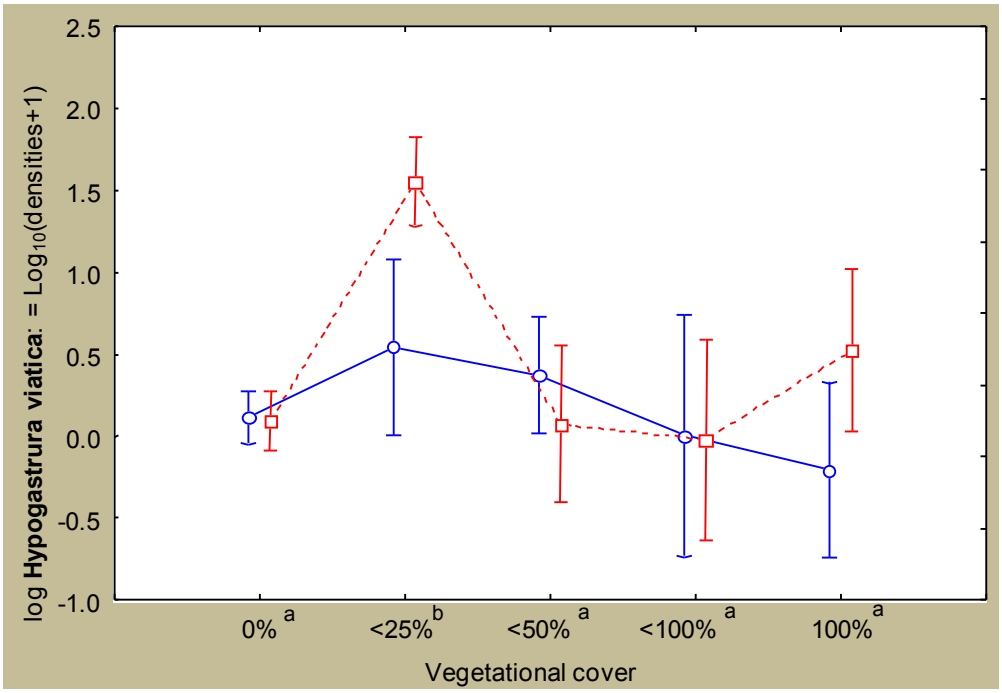


Fig. 76: Results of the covariance analysis (ANCOVA) of the densities of *H. viatica* recorded in 2011 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Protaphorura fimata (Gisin, 1952) and *Folsomia candida* Willem, 1902

Both species are cosmopolitan and prefer biotopes influenced by human activity. They were recorded once on Deception Island (Greenslade & Wise, 1984) in collections made under whale bones on Whalers' Bay, which is geothermally warmed. These species were not recorded by us. The subsequent survival of these species in this area is doubtful. Obviously, these species cannot

penetrate into the climatically more severe environments of the island. See also the remarks to the island in the Discussion. The two records known for *P. fimata* in Antarctica are shown in Fig. 77. Thus far *F. candida* has only been found on Deception Island, but its presence in flower pots inside houses of polar stations in the Arctic and Subantarctic is highly probable.

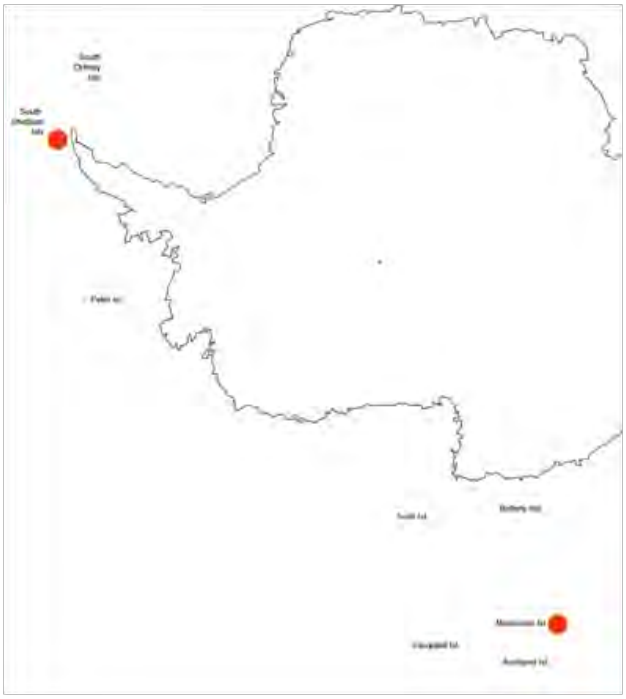


Fig. 77: Records of *P. fimata* in the Antarctic.

Mesaphorura macrochaeta Rusek, 1976

This species was first described from Canada. It is widely distributed in the Northern Hemisphere, being one of the most ubiquist species. It has already been recorded from East Antarctica (Mawson Polar Station) as an exotic species by Greenslade (1992), with the not totally clear note in "pot plant soil" (inside a communication centre, pers.com. Penelope Greenslade), so it cannot be considered to be a true exotic species inhabiting the Antarctic. It was said to be an exotic species on Macquarie Island (Greenslade, 2006) and was recorded once in destroyed greenhouses. It was recorded in one location (Deception Island) in the present studies both in 2010 and 2011. Thus far, the discovery here of several specimens of this species in natural biotopes of Deception Island is at least the first report of the species for the Maritime Antarctic. All records of known distribution are given in Fig. 78.

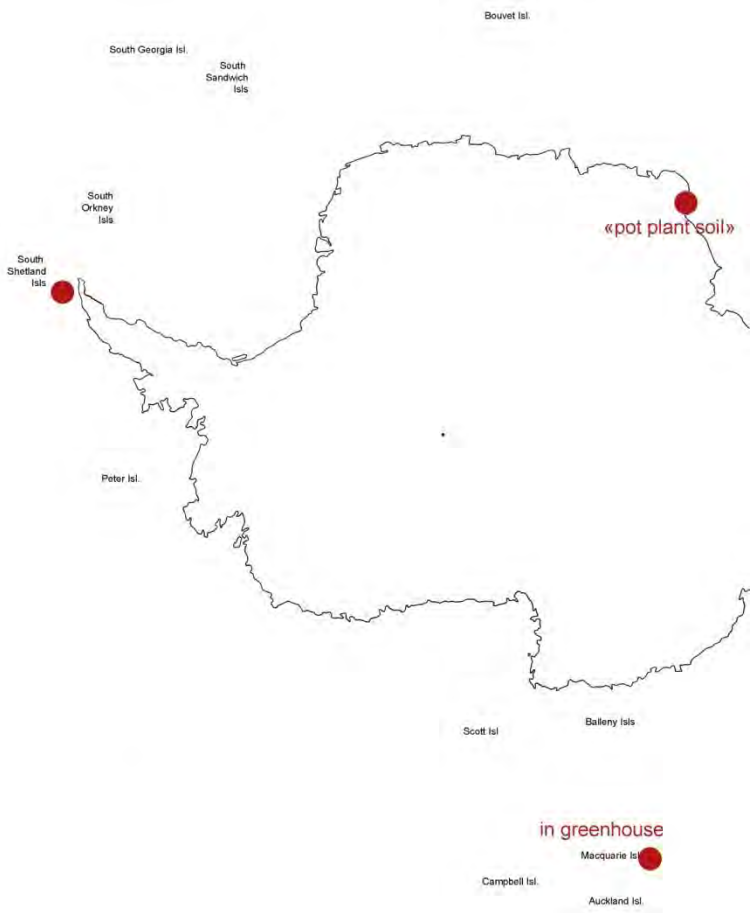


Fig. 78: Records of *M. macrochaeta* in the Antarctic.

M. macrochaeta did not correlate with any habitat factor in either study year. At Deception Island it was generally found in much higher densities in human influenced areas. Due to high sample-to-sample variability, this was statistically significant only in the year 2010 or when both years were analyzed together (Fig. 79; Appendix 5, Table A5-4).

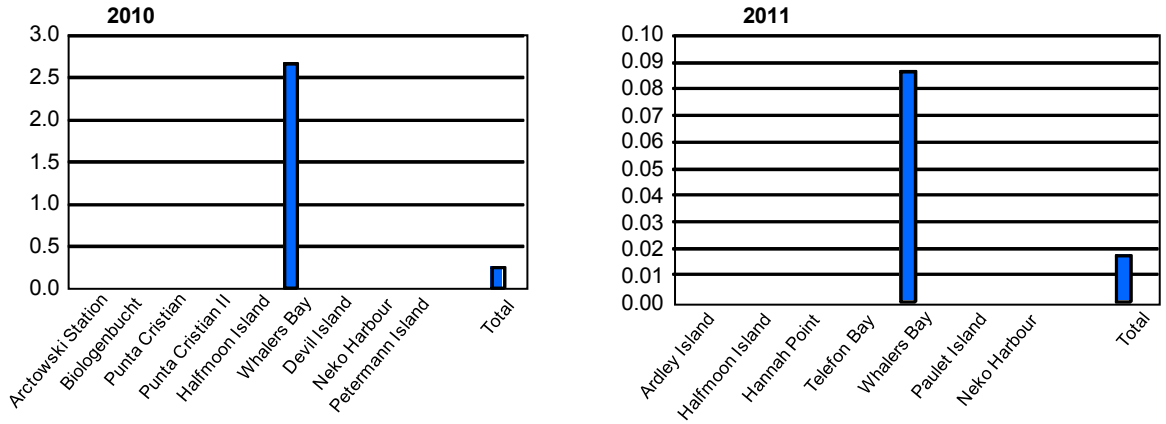


Fig. 79: Total densities of *M. macrochaeta* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analyses showed that in 2010 *M. macrochaeta* occurred in significantly higher densities where no vegetation was present (Fig. 80; Appendix 6, Table A6-1).

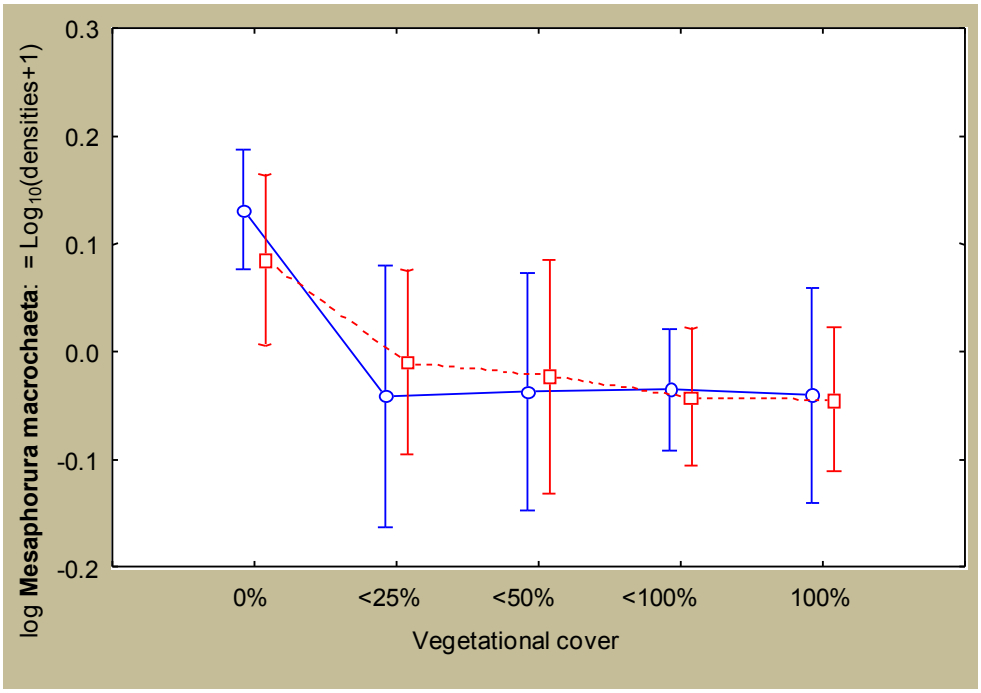


Fig. 80: Results of the covariance analysis (ANCOVA) of the densities of *M. macrochaeta* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

Proisotoma minuta Tullberg, 1871

This species was first described from Europe and is practically cosmopolitan species. It was recorded in the present study once as one individual in 2010. It has already been recorded in the Subantarctic (Fig. 81). It is an exotic species in Macquarie Island (Greenslade, 2006), where it was recorded once in demolished greenhouses.

Due to only one individual being recorded in the present study, no analysis of its relation to habitat factors or human influence on his densities could be analyzed.

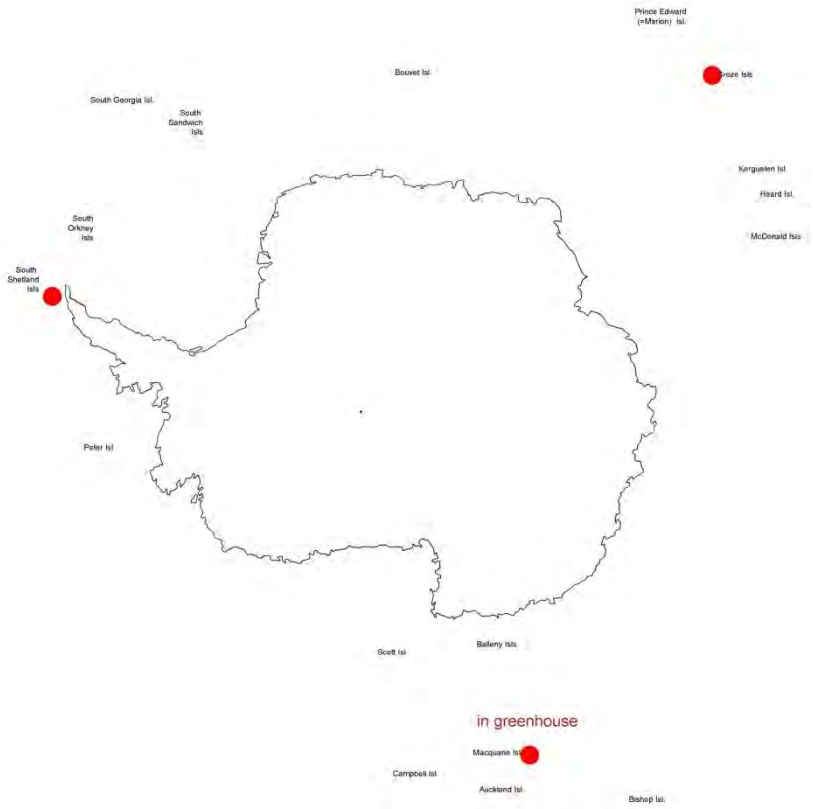


Fig. 81: Records of *P. minuta* throughout the Antarctic and Subantarctic.

Deuteraphorura cebennaria (Gisin, 1956)

D. cebennaria was collected by the British Antarctic Survey and subsequently transferred to the taxonomists of the current project for identification. We accept the taxonomic understanding of the species proposed by Fjellberg (1998) and Pomorski (1998). The species was first described from Europe, where it seems to be widely distributed. Its presence in other parts of the world is less understood due to the ambiguous taxonomy of the group. Formally, our record is the first record of the species in the Southern Hemisphere (more detailed information is given below).

Cryptopygus caecus Wahlgren, 1906

This species was recorded in one location (Whalers Bay on Deception Island) in the current investigation. It is widely known from the Subantarctic and even from southern areas of South Africa, Australia, New Zealand and South America. It was first described from South Georgia (Subantarctic). Deception Island (South Shetland Islands) is the only Antarctic record for this species. It has been found there several times (Tilbrook 1967, Wise 1971, Greenslade & Wise 1984), but not on other neighbouring islands. The same local "thermophilous" distribution was confirmed in the present study. Unlike other non-indigenous species, *C. caecus* is not a cosmopolitan species and its presence in the maritime Antarctic is quite possible. The distribution of the species can, however, become wider when considering *Cryptopygus garretti* Bagnall to be its junior synomym. *Cryptopygus garetti* is a European species, but may also possibly be introduced to Europe (Potapov, 2001). The role of *C. caecus* could be re-estimated even in Subantarctica and the status of this species (native/exotic) is in doubt (see also comments to the Deception Island in the Discussion). Its known records are shown in Figs 82 and 83.

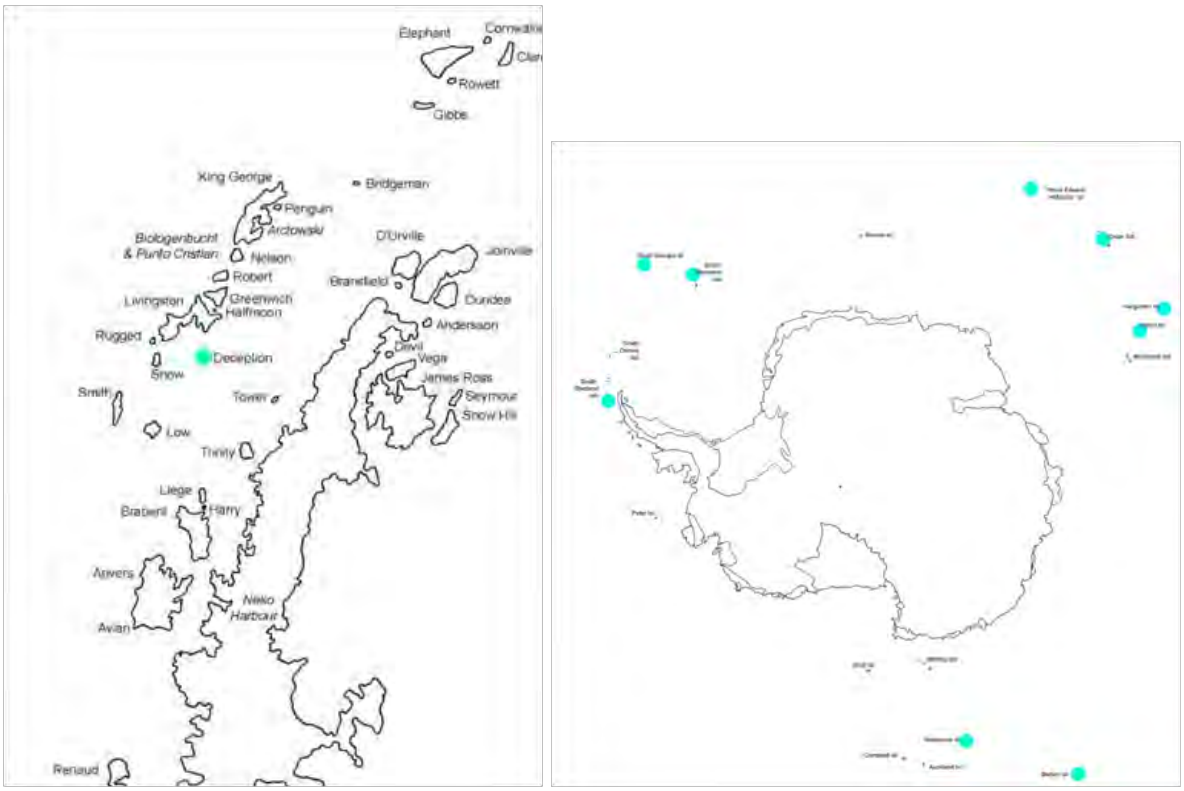


Fig. 82 (left): Record of *C. caecus* in the maritime Antarctic.
Fig. 83 (right): Records of *C. caecus* throughout the Antarctic and Subantarctic.

As in *H. viatica* and in contrast to almost all other collembolan species, *C. caecus* correlated positively to soil temperature and negatively to quantities and quality of soil organic matter (Appendix 4, Table A4-1). Once again, since this species was only recorded in Deception Island in this study, this most likely reflects the conditions of this island more than true habitat preferences of the species.

Regarding its distribution in human influenced and non-influenced areas, *C. caecus* showed contradictory results the two study years (Fig. 84). In 2010 it was more abundant in anthropogenically influenced areas, while in 2011 it was found in somewhat higher densities in the non-influenced areas, albeit in much lower total densities than in the year before. Only the results for 2010 were statistically significant (Appendix 5, Table A5-4).

The covariance analyses confirmed the results of the variance analyses for the year 2010, when significantly higher densities of *C. caecus* occurred in an areas not influenced by humans (Fig. 85; Appendix 6, Table A6-1).

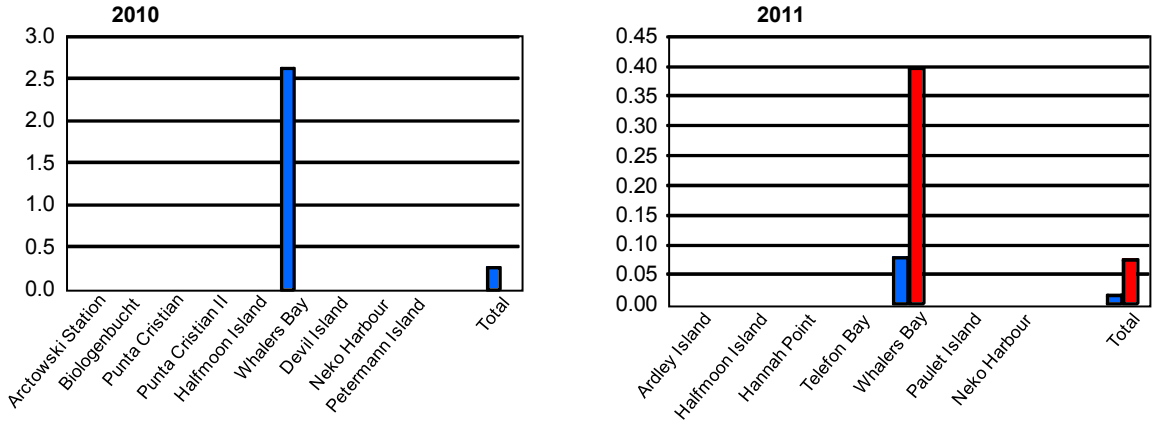


Fig. 84: Total densities of *C. caecus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

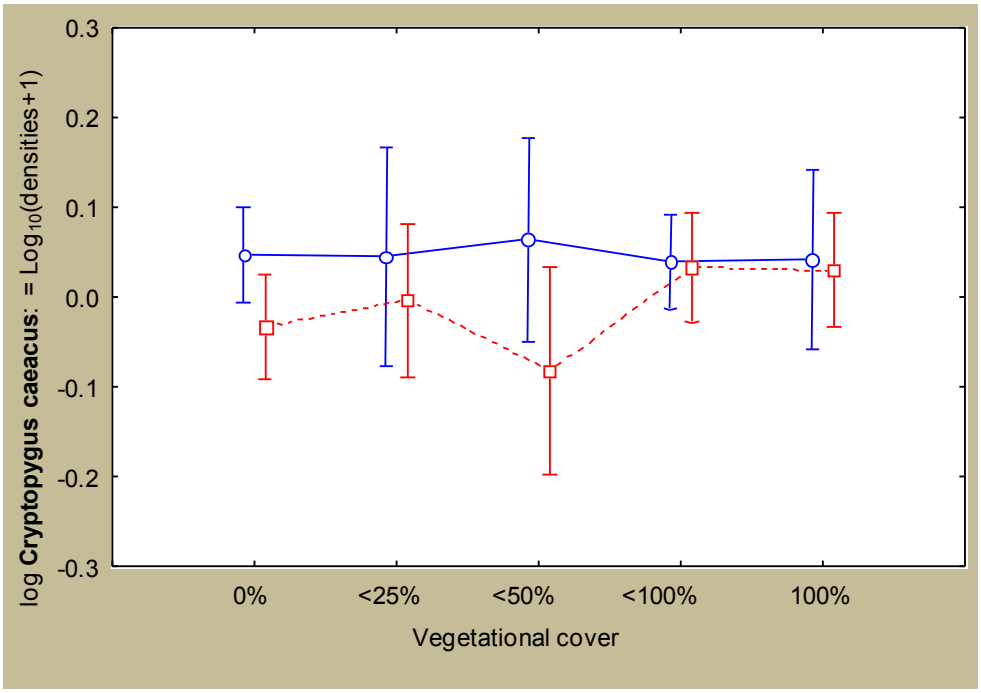


Fig. 85: Results of the covariance analysis (ANCOVA) of the densities of *C. caecus* recorded in 2010 after filtering out various background habitat parameters. Figure explanation as in Fig. 38.

3.4.2 Acari (mites)

3.4.2.1 Actinedida

3.4.2.1.1 General community parameters

A total of almost 2100 individuals of Actinedid mites were registered, over 1600 in the year 2010 and more than 450 in 2011. These mites were found in densities between zero (i.e., on Devil island 2010) and almost 50 individuals per 100 cm^3 substrate (i.e., in various localities of the Fildes Peninsula in 2010). In both years, highly significant differences between localities were determined (see Appendix 5, Table A5-1 for results of the statistical analyses), whereby particularly the densities found in the localities on or around King George Island were significantly higher in both years than those of the remaining localities (Fig. 86).

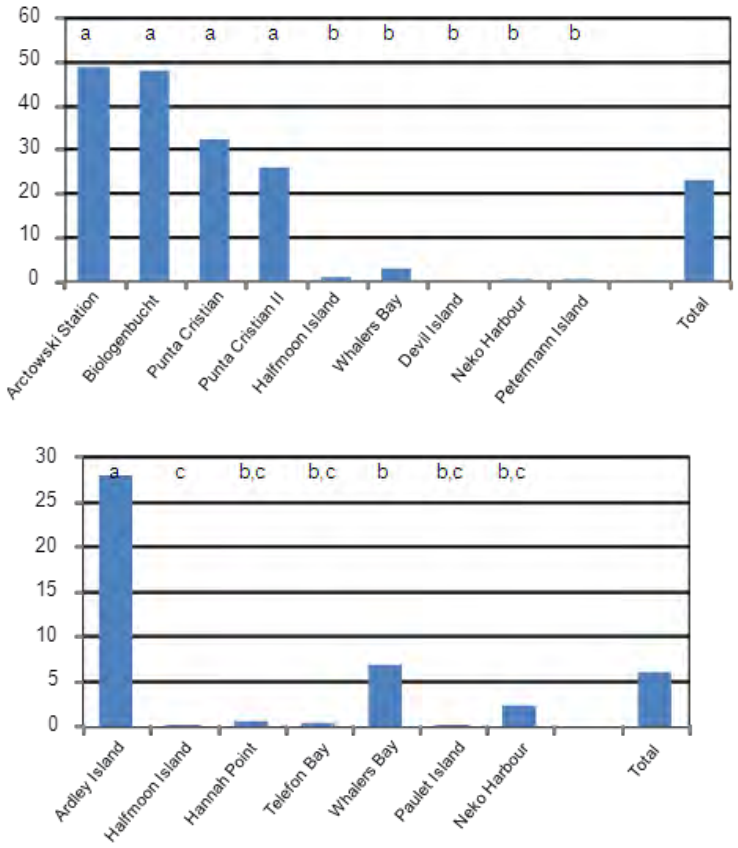


Fig. 86: Total densities of the Actinedida (in individuals per 100 cm^3 substrate) recorded in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

Although almost 4 times as many individuals were detected in the study year 2010 than in 2011, this was not true in those localities that were sampled in both years. In Whalers Bay (Deception Island) and Neko Harbour (Antarctic Peninsula) even the opposite was true, with considerably more individuals having been found 2011 than 2010. On the other hand, on Halfmoon Island somewhat higher densities were found 2010. Due to these contradictory yearly differences among the localities, overall yearly differences throughout all these sites

were statistically not significant (Fig. 87; Appendix 5, Table A5-2). Due to the high sample-to-sample variability, these yearly differences are also not significant within individual localities, whereby only those of Neko Harbour showed a statistical.

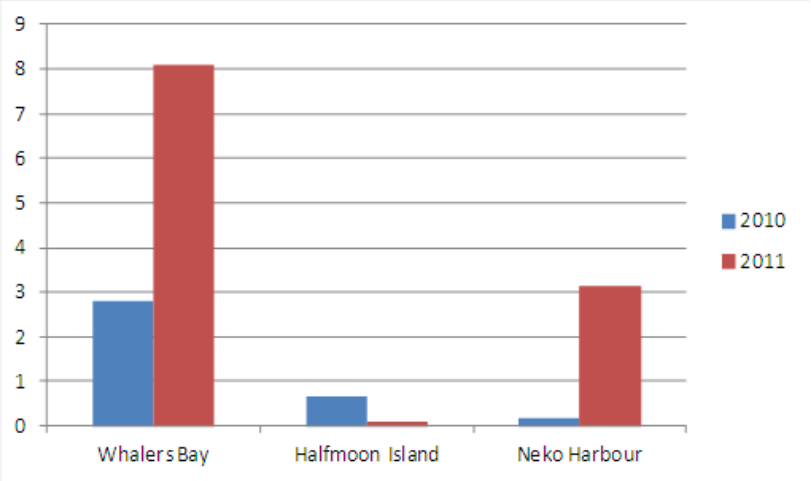


Fig. 87: Total actinedid densities (in individuals per 100 cm^3 substrate) recorded in 2010 and 2011 in the localities studied in both years.

Densities correlated with various abiotic parameters (Appendix 4, Table A4-2). Within both sampling years, densities correlated with location, vegetational cover and soil moisture, although positive relationships between location and vegetational cover were stronger in 2010. Interesting were contradictory relationships between the two study years regarding soil temperature and the various parameters concerning soil organic material; densities correlated negatively to soil temperature in 2010 but positively in 2011, while a positive relationship to amounts and quality (C/N ratio) of organic matter was discernible in 2010 and a negative relationship to amounts of N and C in 2011.

Regarding anthropogenic influence, somewhat higher total Actinedid densities were found in the influenced areas than in the non-influenced areas (Fig. 88). However, this was not true in all localities; the overall differences between influenced and non-influenced areas were therefore statistically not significant (Appendix 5, Table A5-5).

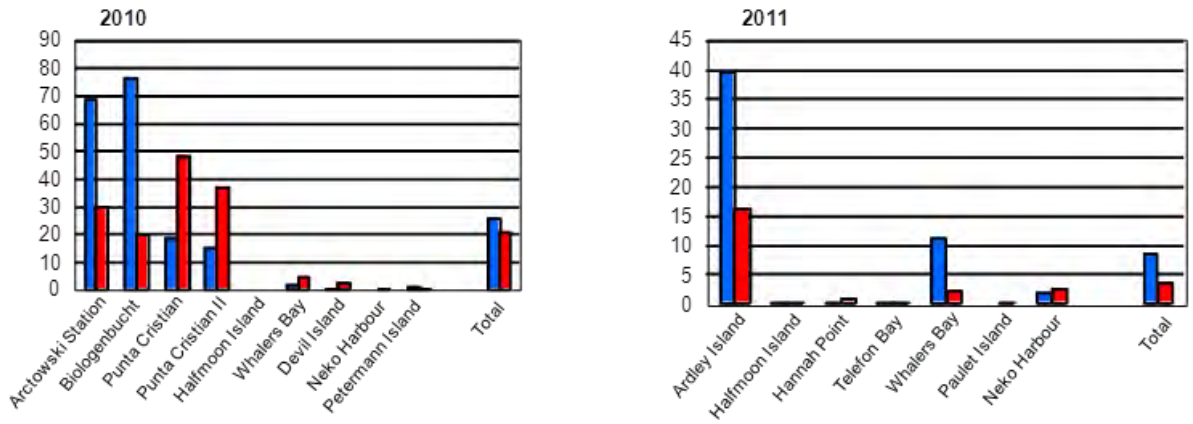


Fig. 88: Total densities of the Actinedida (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Corresponding to the strong correlations to vegetation cover in 2010, the covariance analysis showed a highly significant relationship between total Actinedida densities and vegetational cover (Fig. 89; Appendix 6, Table A6-2). As mentioned in the Methods section, due to statistical difficulties with the data, a lack of its statistical significance within the covariance analyses does not imply negative results. Thus only positive (statistically significant) results are shown here.

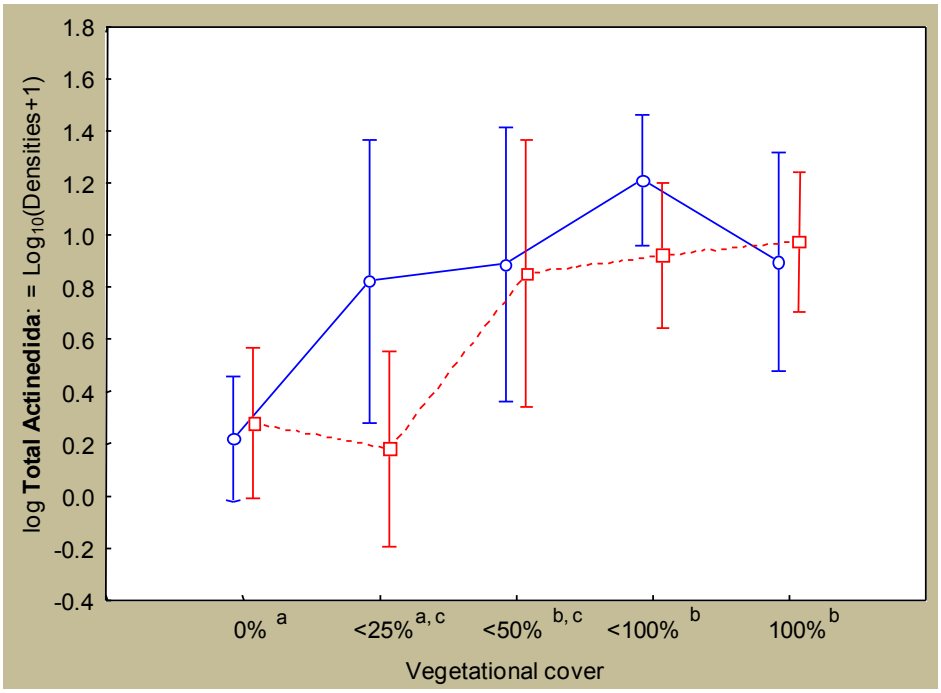


Fig. 89: Results of the covariance analysis (ANCOVA) of total actinedid densities recorded in 2010 after filtering out various background habitat parameters. Densities (given in log individuals per 100 cm^3 substrate) in anthropogenically influenced areas in blue and in non-influenced areas in red. Different letters denote significant collembolan density differences between vegetational cover categories.

A total of 25 separate taxa could be proven, 22 in year 2010 and 18 in 2011. As in the densities, differences in species richness (average species number per area) between individual localities were also highly significant (Appendix 5, Table A5-1), whereby again generally higher species richnesses were found in the localities on King George Island than in the other localities (Fig. 90). However, the highest number of registered species (both years taken together: 17) was found on Whalers Bay, whereby in the other localities between zero (again on Devil Island) and 10-13 total taxa (the localities on the Fildes Peninsula) could be determined.

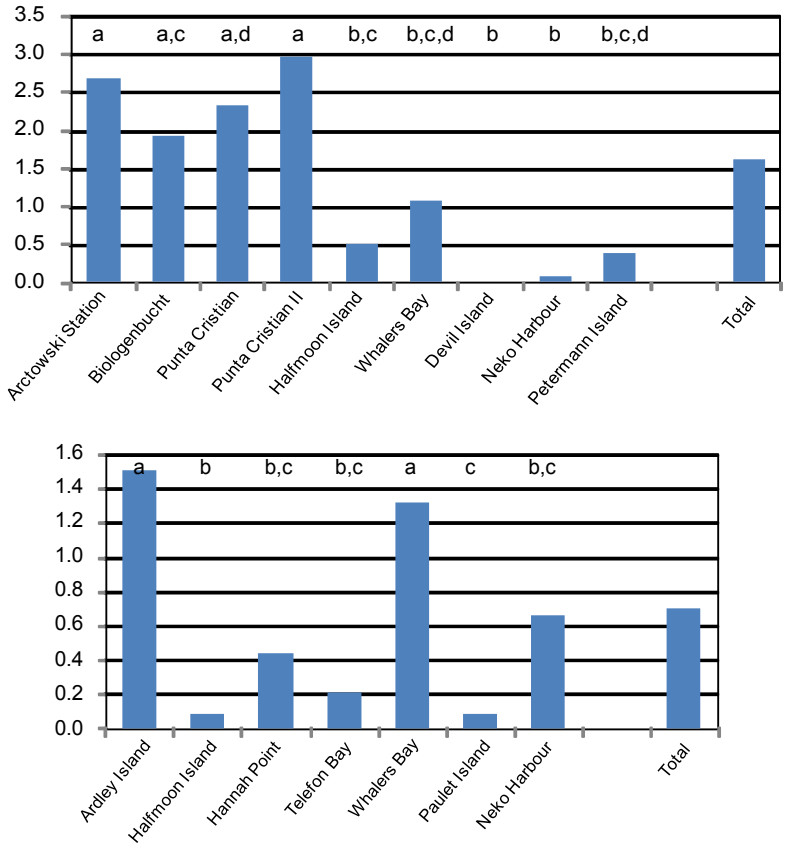


Fig. 90: Total actinedid species richness (in average species per area) observed in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

Although some more taxa could be determined in 2010 than 2011, average species numbers per area in those localities sampled both years often are often higher 2011 (Fig. 91). However, these differences were statistically not significant. Again Halfmoon Island was an exception, where a higher average species richness was observed in the year 2010. As in the densities, within individual localities a statistical tendency for yearly difference was only determined in Neko Harbour.

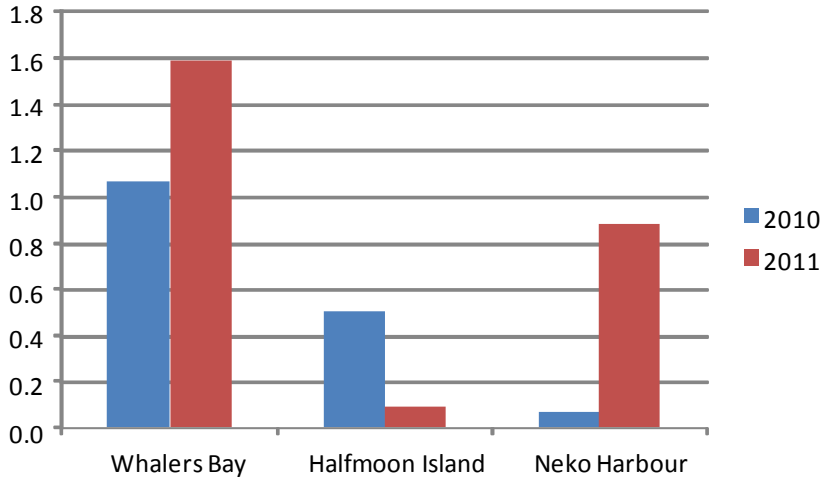


Fig. 91: Species richness of the Actinedida (in average species per sample) registered in 2010 und 2011 in those localities studied in both years.

Correlations between species richness and habitat parameters generally paralleled those of densities (Appendix 4, Table A4-2). Higher species richness correlated with locality as well as increases in vegetational cover and soil moisture. In 2010 species richness correlated negatively with soil pH, meaning more species were found at lower pH values. The contradictory correlations to soil temperature and organic material were also found with species richness.

Again, in total, species richness was slightly higher in the anthropogenically influenced areas than in the non-influenced areas. An opposite difference was, however, observed in some localities (Fig. 92), so that the overall results were again statistically not significant (Appendix 5, Table A5-3).

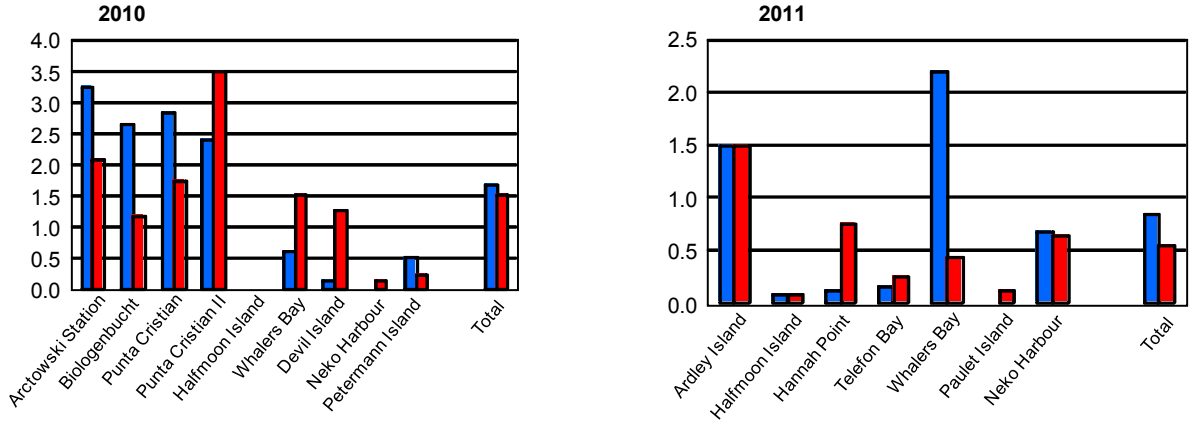


Fig. 92: Total actinedid species richness (in average species number per area) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

As in total densities, in 2010 the covariance analyses also demonstrated a highly significant relationship of species richness with vegetational cover (Fig. 93; Appendix 6, Table A6-2). In contrast to the pure variance analysis (Fig. 92), filtering out various habitat parameters within the covariance analysis revealed higher species richnesses in the anthropogenically influenced areas, which were statistically just significant. These higher densities, however, were mostly found at higher vegetational cover, as elucidated in a significant statistical interaction between human influence and vegetational cover ($F = 2.965$, $P = 0.022$).

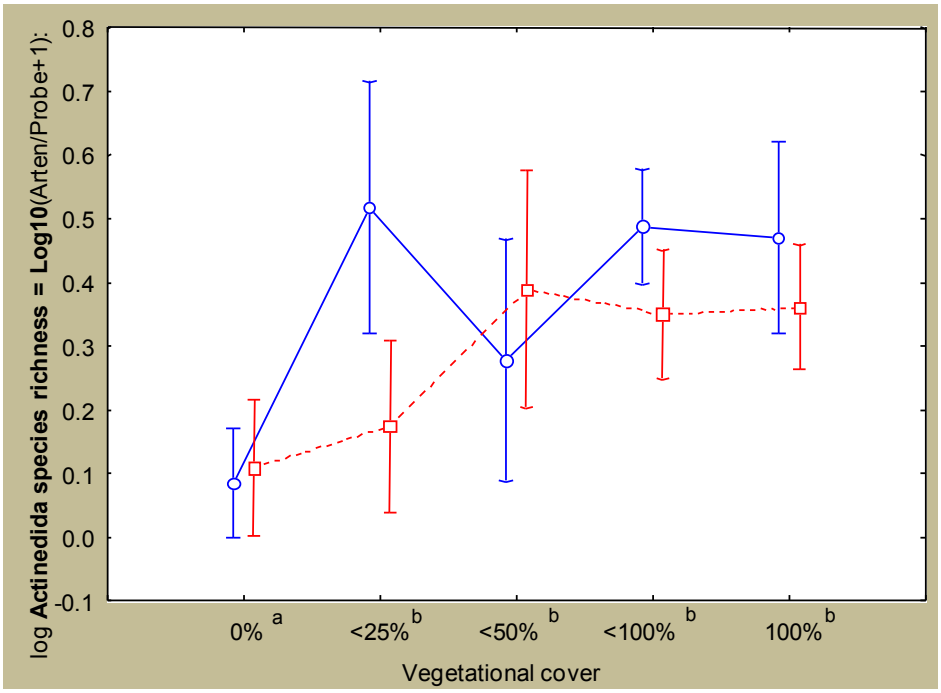


Fig. 93: Results of the covariance analysis (ANCOVA) of the species numbers of Actindida recorded in 2010 after filtering out the influence of various background parameters. Figure explanation as in Fig. 89.

3.4.2.1.2 Results and description of the determined species

The species determined in the present project are given in their systematical position in the follwing. Their average densities in the various localities are given in Appendix 3, Table A3-2.

Endeostigmata

Alycoidea

Alicorhagiidae

Alicorhagia Berlese, 1910 sp.

Nanorchestidae

Nanorchestes cf. *anarcticus* Strandtmann, 1963
Nanorchestes berryi Strandtmann, 1982
Nanorchestes nivalis (= gressetti) (Strandtmann, 1982)
Nanorchestes cf. *lalae* Strandtmann, 1982
Nanorchestes marianae Strandtmann, 1982
Nanorchestes n. sp. [nah *brekkeristae* Strandtmann & Sømme, 1977]
Nanorchestes sp. V
Speleorchestes Trägårdh, 1909 sp.

Terpnacarida

Terpnacarus gibbosus (Womersley, 1944)

Prostigmata

Eupodina

Eupodoidea

Eupodidae

Eupodes (*Protereunetes*) *minutus* (Strandtmann, 1967)
Eupodes (*Protereunetes*) *exiguus* Booth, Edwards & Usher, 1985
Eupodes (*Protereunetes*) *parvus* ssp. grahamensis Booth, Edwards & Usher, 1985

Penthalodidae

Stereotydeus villosus (Trouessart, 1902)

Rhagidiidae
Rhagidia gerlachei (Trouessart, 1903)
Rhagidia Thorell, 1871 sp. juv.

Tydeoidea
Ereynetidae
Ereynetes (*Gymnereynetes*) *macquariensis* Fain, 1962

Meyerellidae
Apotriophtydeus cf. *wilkesi* (Strandtmann, 1967)
Apotriophtydeus scotia Usher & Edwards, 1986
Pretriophtydeus tilbrooki (Strandtmann, 1967)

Iolinidae
cf. *Coccotydaeolus krantzi* Baker, 1965

Tydeidae
Lorryia Oudemans, 1925 sp.

Eleutherengona
Rhaphignathae
Raphignathoidea
Stigmaeidae
Gen. sp. juv.

Eriophyoidae
Eriophyidae
Gen. sp. juv.

Heterostigmata
Pygmephoroidea
Pygmephoridae
Bakerdania cf. *antarcticus* (Mahunka, 1967)

Tarsonemoidea
Tarsonemidae
aff. *Tarsanonychus* Lindquist, 1986 sp.
Tarsonemus s.s. Canestrini & Fansago 1876 sp.
Gen. sp. juv.

3.4.2.1.2.1 Species native to Maritime Antarctica

In the following individual species known to occur in the Antarctic are described together with information on their known distribution and ecology. Species new and potentially non-native to the Antarctic are described in the next section. Actinedid mites have been fairly regularly studied in the maritime Antarctic in the past decades, particularly by members of, e.g., the Bishop Museum (Honolulu) or the British Antarctic Survey. The studied sites were widespread throughout the Antarctic Peninsula, as far south as 71° S (Alexander Island), albeit particularly on the west side of the Peninsula and in localities assessable from various research stations (Fig. 94). Although the studied localities in the present project were concentrated in the northern half of the Antarctic Peninsula, with some localities visited that had been studied by other research groups in the past, most of the studied sites represent new additions to the known distribution of the Antarctic Actinedid fauna (Fig. 94).

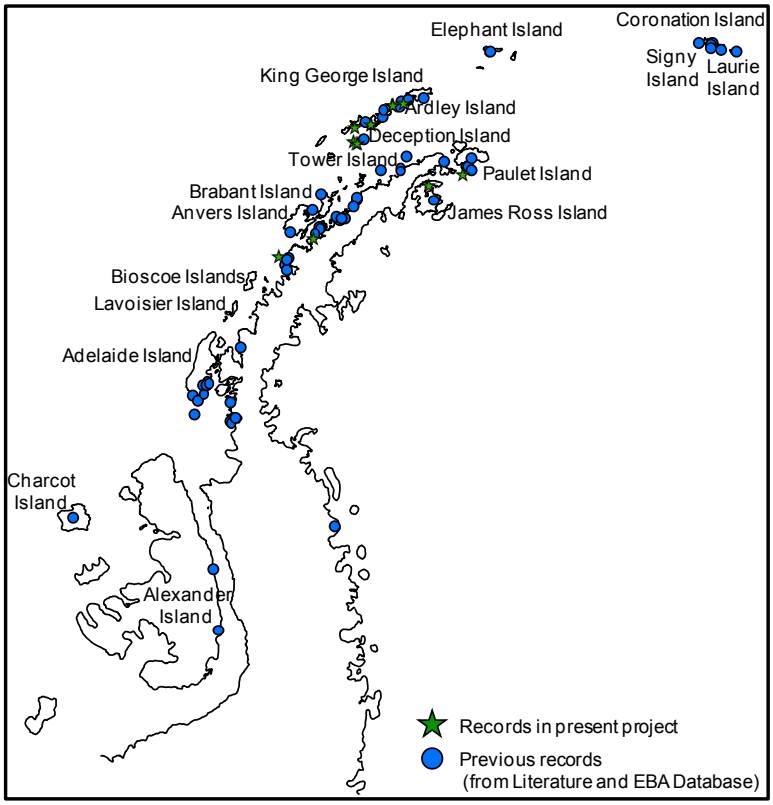


Fig. 94: Previous (blue points) and present records (green stars) of the actinedid species registered in the present project. Sources: acarological and soil faunal literature from the Antarctic (as far as available to the authors) as well as the Biodiversity Database of the Australian Antarctic Data Centre.

Nanorchestes

The genus *Nanorchestes* is one of the most common Actinedid genus found throughout the world in a plurality of various habitat types. In the Antarctic the genus is widespread and presently 14 species of this genus are known, the majority of which are only known from this continent (Pugh 1993). The genus also occurs widely in the Arctic, but the species found in the two poles are generally very different (Strandtmann 1968). The taxonomy of Antarctic *Nanorchestes* underwent a strong revision in the 1980s, with many new species described and the identity of previously determined species of this genus proven highly questionable. Records of *Nanorchestes* from the maritime Antarctic previous to this time usually referred to *N. antarcticus*, of which there are no longer any verified records in this area (Convey & Quintana 1997, Convey et al. 2000a). In the present study seven species of this genus were determined, only two of which (*N. berryi* and *N. nivalis*) could be determined with absolute certainty. Three species (*N. antarcticus, N. lalae, N. marianae*) are only known from continental Antarctica and, due to only few individuals of these species having been registered and taxonomic uncertainties, their determination here must still be considered tentative. If their identities prove true, then this would be their first proven occurrence in the maritime Antarctic. Two other species could not be identified with the available literature and may possibly represent undescribed species.

This genus is found in a wide variety of different habitats in the Antarctic, in moss patches, lichens, soils rich in organic matter, algae (*Prasiola crispa*), the littoral zone and often in large concentrations under stones and rocks (Gressitt 1967, Goddard 1979b, Usher & Booth 1984). In

moss turf it is generally found in surface layers, often in large aggregations, whereby juveniles can be found in deeper layers (Goddard 1979a, Usher & Booth 1984). The various species of this genus are all considered to feed on red and green algae as well as partly also on fungal hyphae (Strong 1967, Fitzsimons 1971, Goddard 1979b, 1979c, Convey & Quintana 1997).

Members of the genus show a wide tolerance for various environmental conditions, e.g., being active between -20°C and +25°C and showing higher tolerances to lower humidities than other Actinedid species and perhaps being the only mite species capable of surviving in barren chalikosystem habitats (Goddard 1979b). It cannot be determined if this wide range of tolerances is true of all species of this genus or due to the studies being undertaken on various species, which was unknown at the time. This taxon generally has very fast developmental rates and thus can quickly develop individual-rich populations (Usher & Booth 1986). The average generation time is considered to be two years, whereby all developmental stages can overwinter (Usher & Booth 1986).

Nanorchestes nivalis (= gressetti) (Strandtmann, 1982)

N. nivalis, previous to Judson (1995) known as *N. gressetti*, is the most widespread species of this genus in the maritime Antarctic (Fig. 95). For instance, it occurs on all of the South Sandwich Islands, where it was found in 50% of samples taken there (Convey et al. 2000a), which in the Antarctic may be considered very widespread. It is most likely of maritime Antarctic origin (Convey et al. 2000a), but has also been found in the Subantarctic, e.g., South Georgia (Convey et al. 2000b). It has generally been found in mosses and algal mats, often in high densities (Gressitt 1967, Convey & Quintana 1997, Convey et al. 2000b), as well as in green and red algae on snow (Gressitt 1967), but rarely under stones (but see Convey & Smith 1997).

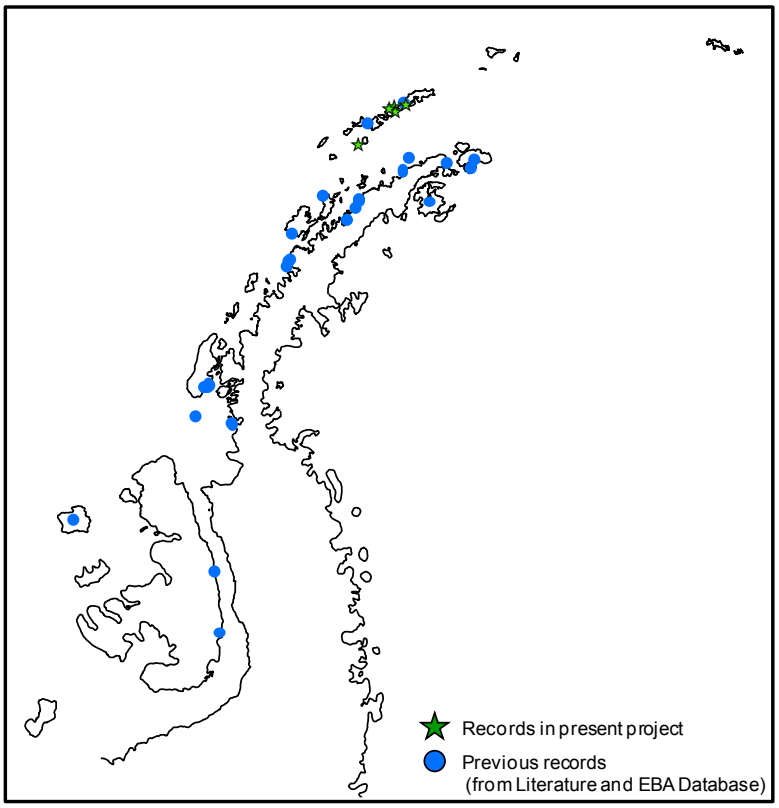


Fig. 95: Previous records of *N. nivalis* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

In the present study the species was only found in 2010 in single to few individuals in individual samples, usually from the vegetated areas in or around King George Island (Fig. 95). It was also found on Whalers Bay (Deception Island), but only in non-influenced areas where a light vegetational cover was present. This species showed no overall correlation to any habitat parameter.

Regarding human influence only the data from 2010 could be analyzed (Fig. 96). In most localities around King George Island, the species was only found in influenced areas; although in the second locality at Punta Christian in both area types with higher densities in the non-influenced areas. However, overall, no significant influence could be determined.

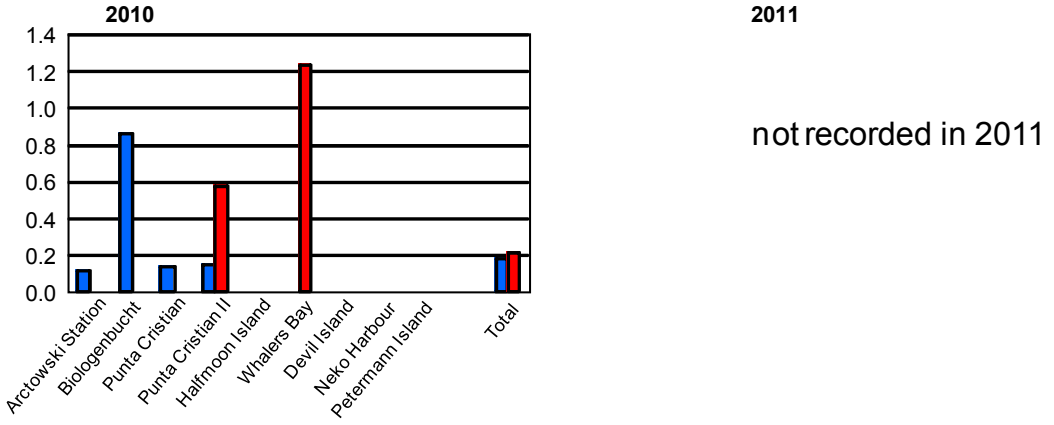


Fig. 96: Total densities *N. nivalis* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Nanorchestes berryi Strandtmann, 1982

N. berryi is also fairly widespread around the Antarctic Peninsula, but has been determined far less often than the previous species (Fig. 97). It has generally been found in vegetated habitats, e.g., lichens, mosses or swards of *Deschampsia antarctica*, but rarely under stones (Usher & Edwards 1984, Convey & Quintana 1997, Convey & Smith 1997). The species has sometimes been associated with dryer habitats than *N. nivalis* (Convey & Quintana 1997).

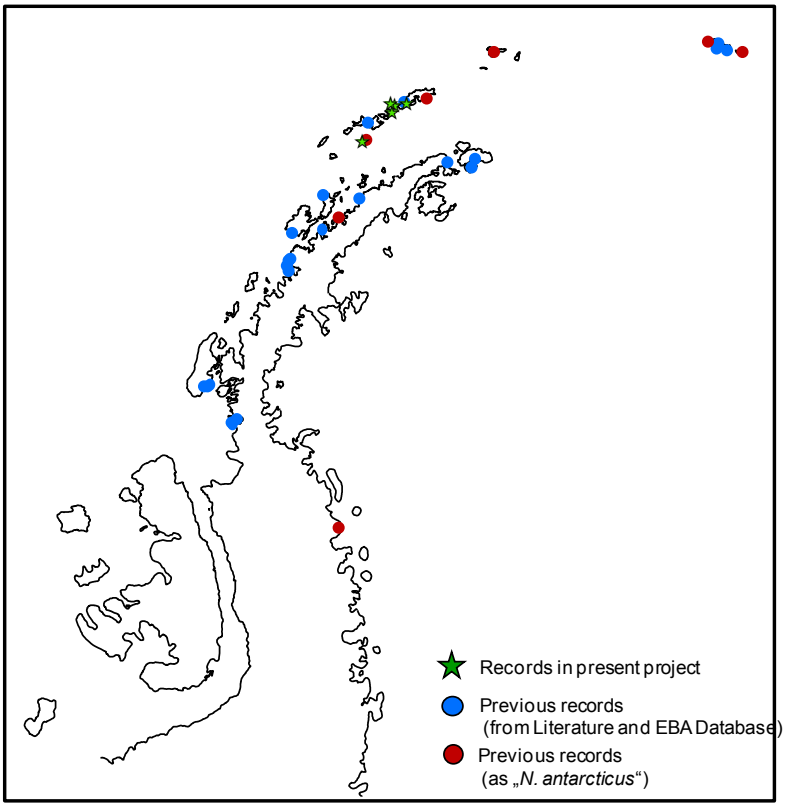


Fig. 97: Previous records of *N. berryi* in the maritime Antarctic (blue dots, records of presumed misidentifications as red dots) and records from the current project (green stars). Sources: as in Fig. 94.

In this study, *N. berryi* was generally found in the same locations as the previous species (cf Figs 95 and 97), but in far higher densities and in both study years. The species was positively correlated to location and vegetational cover in both years as well as to soil moisture and quantity and quality of organic material in 2011 (Appendix 4, Table A4-2). The species' occurrence also correlated to soil temperature, however negatively in 2010 and positively in 2011.

N. berryi occurred in both human influenced and non-influenced areas, whereby quantitative differences between the two area types were often location specific (Fig. 98). Due to these locality-specific results, the differences were statistically only tendencial, but not significant in 2010 (Appendix 5, Table A5-5). However, in 2011 significantly more individuals were found in human influenced areas, which became even more significant when both years were analyzed together.

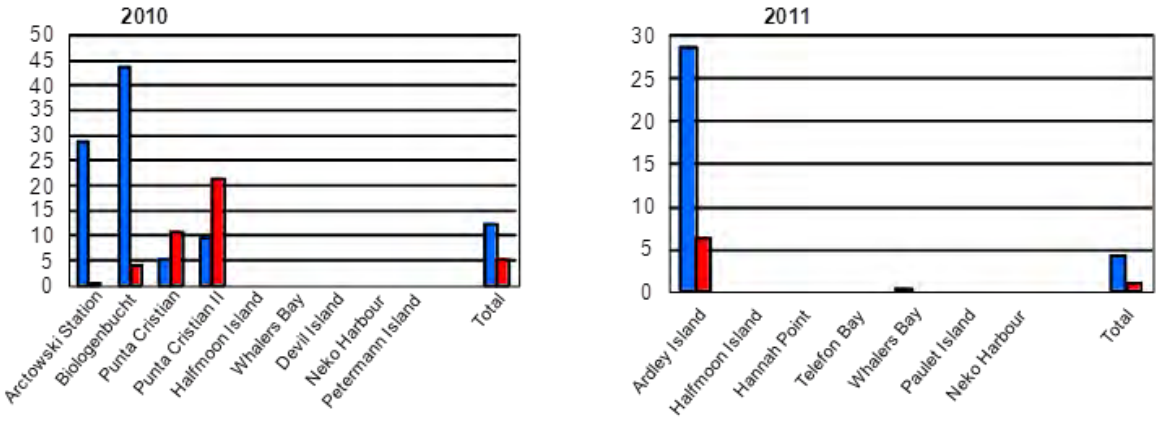


Fig. 98: Total densities *N. berryi* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Interestingly, covariance analyses revealed similar results but in the opposite years to the pure variance analyses. In 2010, filtering out various habitat parameters showed significantly higher densities of *N. berryi* in human-influenced areas (Fig. 99; Appendix 6, Table A6-2), which were larger with increasing vegetational cover (interaction human influence x vegetational cover). In 2011, on the other hand, the covariance analysis primarily revealed a significant relationship between the densities of *N berryi* and vegetational cover (Fig. 100).

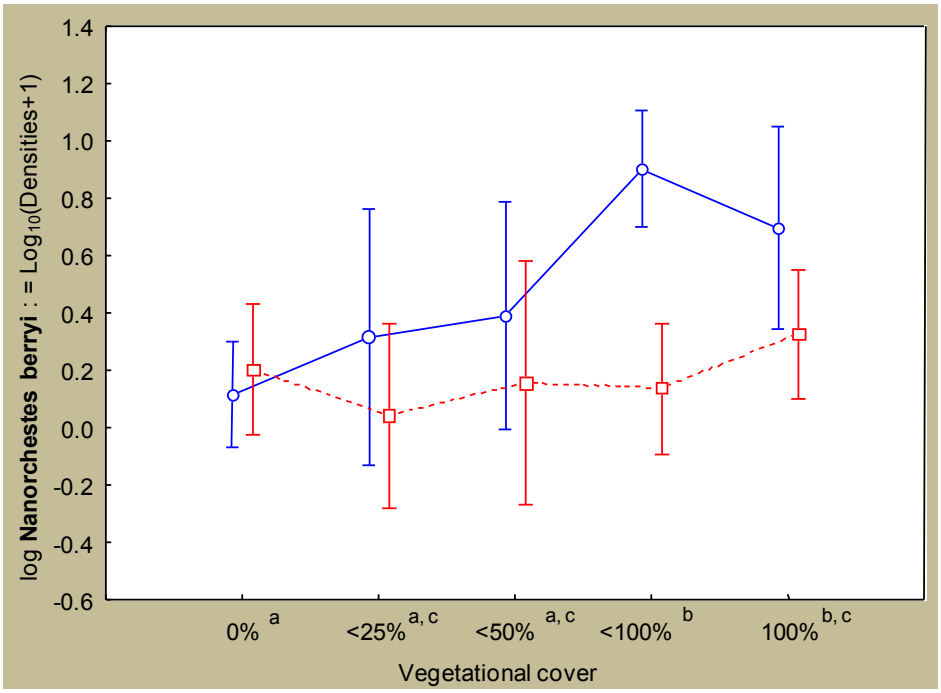


Fig. 99: Results of the covariance analysis (ANCOVA) of the densities of *N. berryi* recorded in 2010. Figure explanation as in Fig. 89.

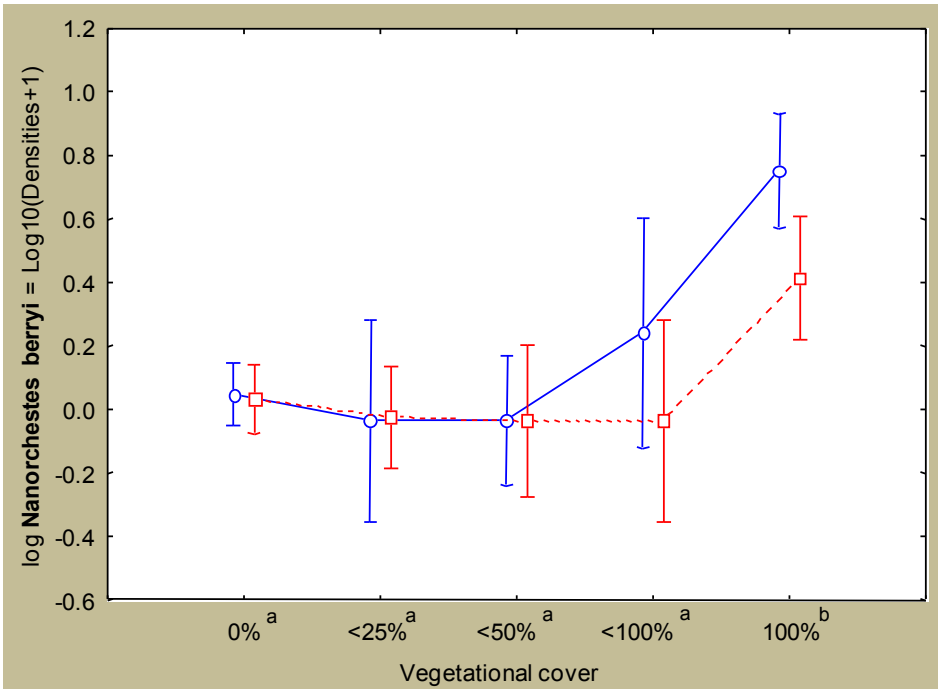


Fig. 100: Results of the covariance analysis (ANCOVA) of the densities of *N. berryi* recorded in 2011. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

Eupodes (Protereuntes)

Eupodes represents another fairly species-rich and widely distributed genus in the Antarctic. This genus is also distributed worldwide in a high variety of habitat types, where it often represents one of the most dominant Actinedid taxa in soils. In the Antarctic at least nine species of this genus are known, all of which only occur in the Antarctic or Subantarctic (Pugh 1993, Booth et al. 1985). The most common species of this genus in the maritime Antarctic belong to the subgenus *Protereuntes*, where three species are known (*E. minutus*, *E. exiguus*, *E. parvus* with 2 subspecies).

The genus has generally been found in moss turf or young moss patches as well as in patches of *Deschampsia antarctica*, often very abundantly, but more rarely under stones (Gressitt et al. 1963, Gressitt 1967, Usher & Booth 1984). It is generally fairly evenly distributed throughout the vegetation profile, whereby adults can be found highly aggregated in surface layers and juveniles can penetrate deeper into the profile (Usher & Booth 1984). Species in this genus have not been found to have a clear yearly cycle; eggs often hatch soon after being laid and all life stages can overwinter; their mean generation time has been determined to be 1-2 years (Usher & Booth 1986). They are assumed to feed on fungal hyphae, algae but may also be a scavenger (Strong 1967, Goddard 1979c).

Besides the three species listed below, many individuals of this genus found in the present study were juvenile, which could not be determined to species level.

Eupodes minutus (Strandtmann, 1967)

The most commonly found and widespread *Eupodes* species in the maritime Antarctic is *E. minutus* (Fig. 101). However, earlier studies in Antarctic only recorded this species (Gressitt et al. 1963, Gressitt 1967, Usher & Booth 1984), which after a taxonomic revision in the mid-1980s

is now known to include the three species listed above (Booth et al. 1985). Therefore, much of the information regarding the species, including the distribution given in Fig. 101, could possibly concern a combination of different species.

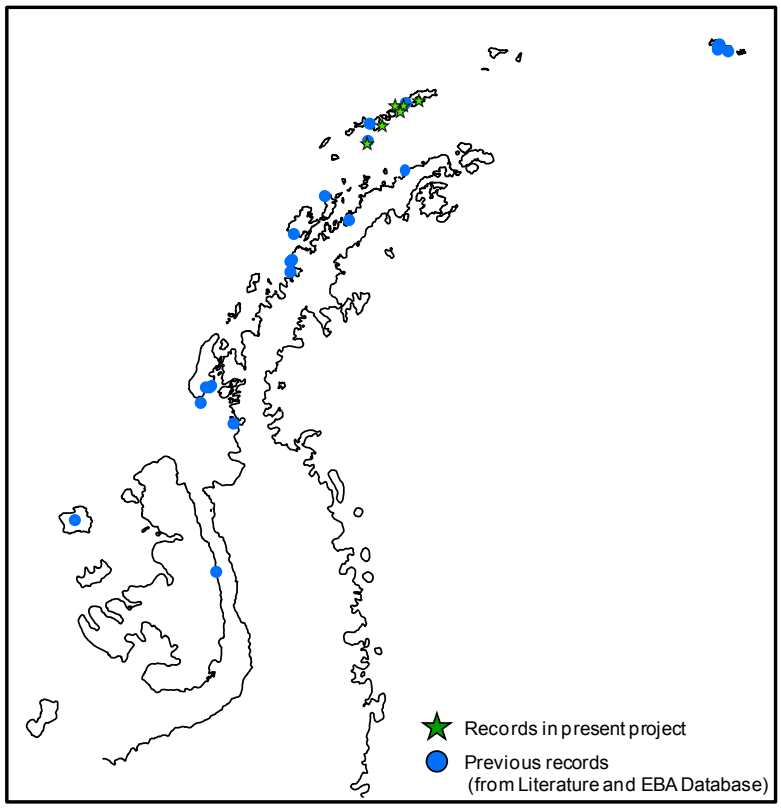


Fig. 101: Previous records of *E. minutus* in the maritime Antarctic (blue dots) and records from the current project (green stars). Sources: as in Fig. 94.

This possibility of species confusion notwithstanding, *E. minutus* has been found widespread in many maritime Antarctic sites as well as in the Subantarctic, e.g., South Georgia (Goddard 1979b), the Macquarie & Prince Edward Islands (Marshall et al. 1999, Barendse 1999, zit in Convey et al. 2000b). The species has been generally found in the upper layers of mosses, *Deschampsia antarctica*, lichens and *Prasiola crispa* mats as well as (rarely) under stones (Goddard 1979a, 1979b, Convey et al. 2000b, Usher & Edwards 1984). It is fairly susceptible to desiccation and avoids dry areas and is photonegative (avoids sunlit areas) (Goddard 1979b). Its main food resource is most likely epiphytic algae as well as fungal hyphae (Strong 1967, Goddard 1979c).

E. minutes was found primarily during this study in the locations on and around King George Island, but also on Halfmoon Island as well as Deception Island (Whalers Bay), but only in 2010 (Fig. 101). Within these localities, however, it was only found in individual samples in only a few individuals, which is in contrast to previous published reports of the species. No significant correlation to any habitat parameter could be found. It was found in both human influenced and non-influenced areas, whereby density differences varied from location to location, so that no statistically significant human influence could be determined for this species (Fig. 102).

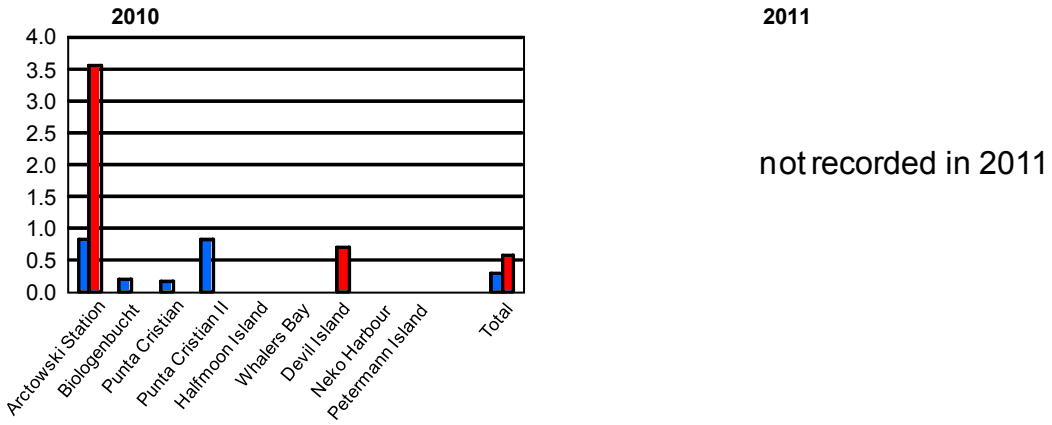


Fig. 102: Total densities *E. minutus* (in individuals 100 cm^{-3}) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010.

Eupodes exiguus Booth, Edwards & Usher, 1985

E. exiguus is a small Eupodid species very similar to *E. minutus*, which apparently has not often been found previously in the maritime Antarctic (Fig. 103). Accordingly, almost no ecological information regarding preferred habitat types or nutritional resources could be found.

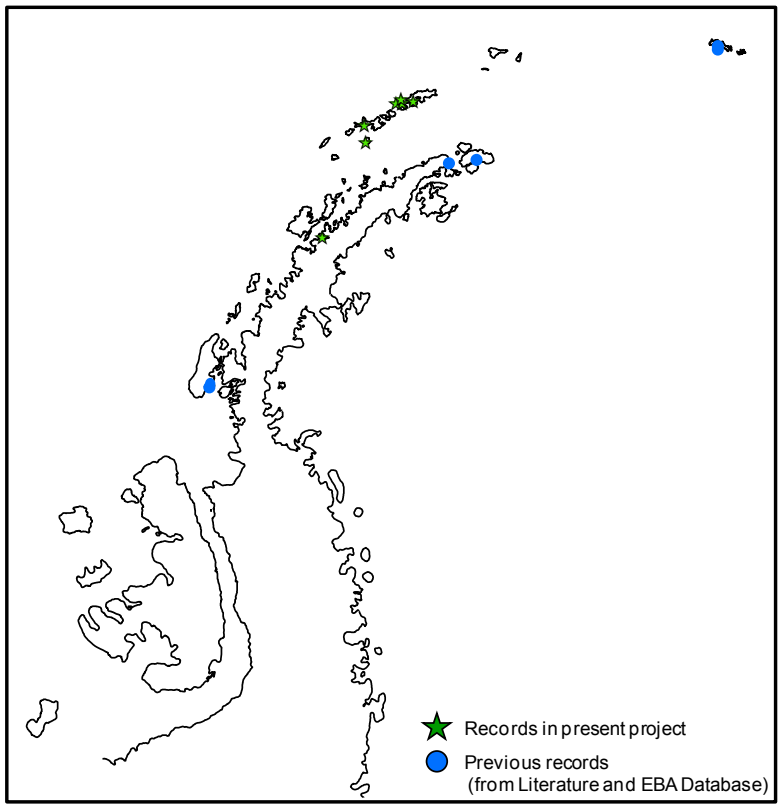


Fig. 103: Previous records of *E. exiguus* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

In contrast to the paucity of published information on *E. exiguus*, it was the most abundant *Eupodes* species found in the present study in both years. It was widespread in the South Shetland Islands, but was also found on the Peninsula itself (Fig. 103). It frequently occurred sympatically (= together) with *E. minutus*, but in densities that were often an order of

magnitude larger. Due to the morphological strong similarity with *E. minutus*, older literature data could possibly result from misdeterminations of *E. exiguus*. However, since the latter species possesses taxonomic characteristics not present in the former species, identification of these characters allows a certain identification.

E. exiguus correlated positively to vegetational cover as well as amounts and quality of organic material and negatively to soil temperature, but only in 2010 (Appendix 4, Table A4-2). This species was found in both human influenced and non-influenced areas, whereby its densities were often significantly higher in human influenced areas (Fig. 104; Appendix 5, Table A5-5).

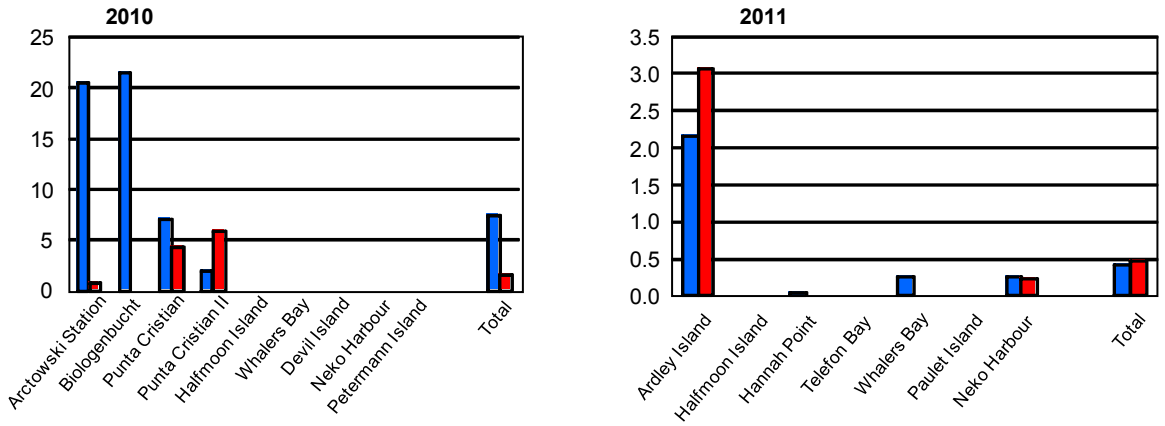


Fig. 104: Total densities *E. exiguus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

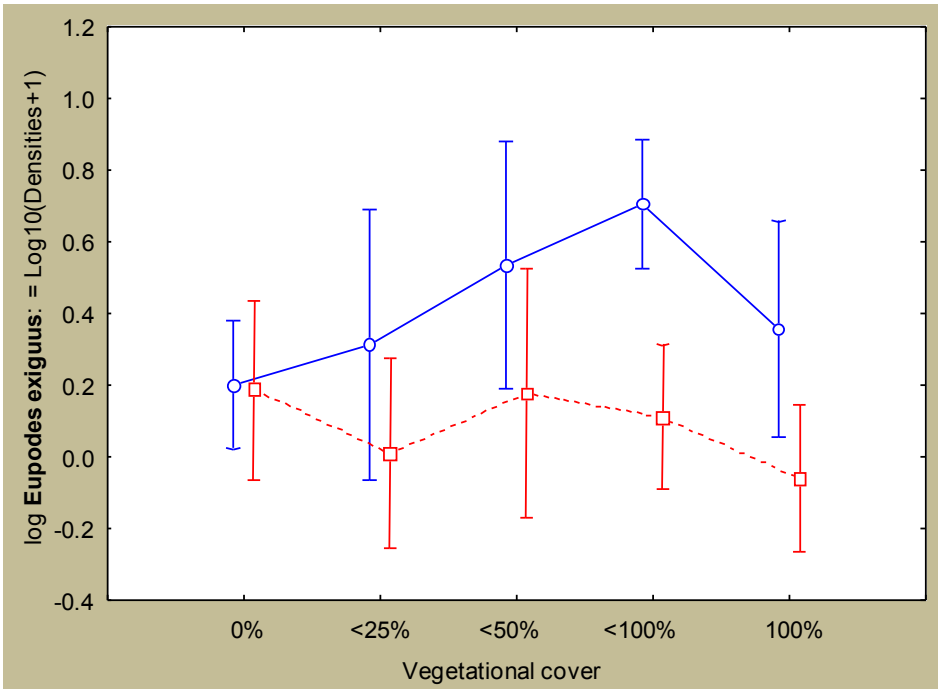


Fig. 105: Results of the covariance analysis (ANCOVA) of the densities of *E. exiguus* recorded in 2010. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

The covariance analysis revealed similar results to the pure variance analysis, whereby densities of *E. exiguus* were significantly higher in human influenced areas, particularly in 2010 (Fig.

1050; Appendix 6, Table A6-2). These differences were stronger with higher vegetational cover, as is apparent in the statistical interaction between human influence and vegetational cover.

Eupodes parvus Booth, Edwards & Usher, 1985

E. parvus is somewhat larger than the two previous species. It has been found fairly widely distributed on and around the Antarctic Peninsula (Fig. 106), more so than, e.g., *E. exiguus*, perhaps due to it being more easily seen. Little explicit information has been given about its habitat preferences other than single observations of it being only found in algal mats, mosses and nests, with his highest densities in dead mosses, but not being found under stones (Convey & Quintana 1997).

In the present study this species was only found on King George Island and Ardley Island (Fig 106), which were the most vegetated localities sampled here. It was usually only observed in a few individuals in sporadic samples. *E. parvus* did not correlate significantly to any habitat parameter. In the year 2010 it was found exclusively in non-influenced areas of King George Island, and therefore its densities in these areas were significantly higher than in human influenced areas (Fig. 107; Appendix 5, Table A5-5). However, in the following year many more individuals were found in anthropogenically influenced areas, albeit exclusively in Ardley Island. The higher densities in influenced areas were partly due to an aggregation of many individuals in a single sample. Due to this high variability, the differences between influenced and non-influenced areas were statistically not significant in the year 2011. Due to these different distributional differences from year to year and between localities, no overall human influence of the occurrence of *E. parvus* could be statistically determined. The covariance analyses did not reveal any significant results.

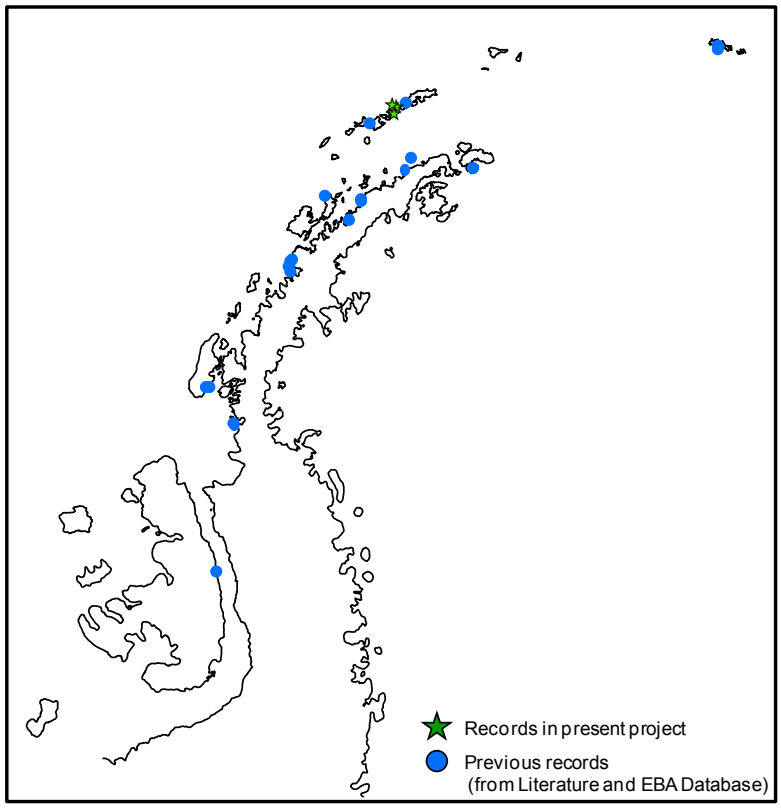


Fig. 106: Previous records of *E. parvus* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

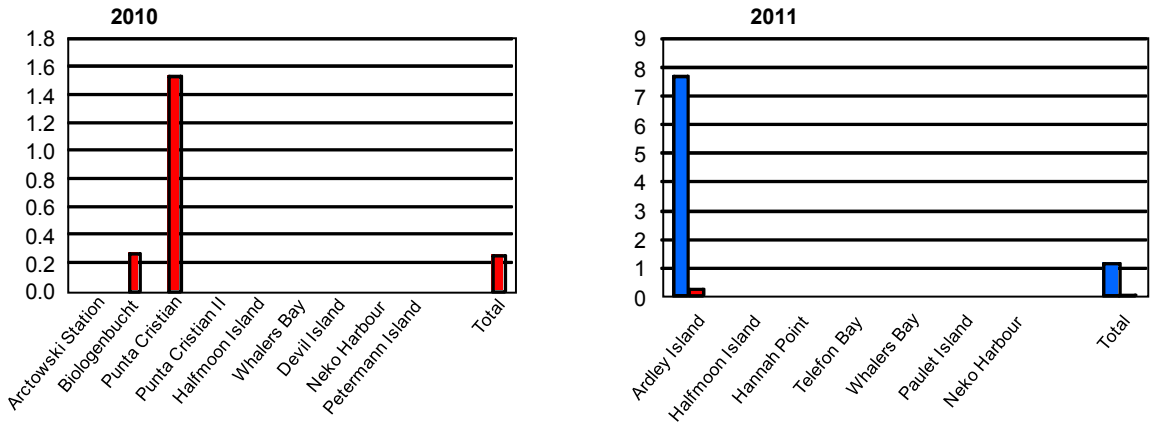


Fig. 107: Total densities *E. parvus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Stereotydeus villosus (Trouessart, 1902)

The genus *Stereotydeus* is also one of the most species-rich and widespread genera in the Antarctic, with eight species occurring on the continent and associated islands (Goddard 1979b, Pugh 1993). Within the family Penthalodidae there is a strong generic difference between the Arctic and Antarctic faunas, with species of the genus *Penthalodes* occurring only in the Arctic (as well as other sites worldwide) and in the Antarctic only species of *Stereotydeus* being present (Strandtmann 1968).

The species *S. villosus* is only known from the maritime Antarctic (Convey et al. 2000b), where it is widespread (Fig. 108) and occurs in many different habitat types (Strong 1967, Gressitt 1967, Convey & Quintana 1997, Convey & Smith 1997). Nonetheless, it has been found most frequently and in higher densities on or under stones or in rocky habitats (Gressitt 1967, Goddard 1979b, Usher & Booth 1984, Richard et al. 1994, Convey & Quintana 1997, Convey & Smith 1997), but also if rarer in mosses, lichens as well as *Deschampsia antarctica* (Dalenius 1965, Gressitt 1967, Usher & Edwards 1984, Convey & Quintana 1997, Gressitt et al. 1963). The species is somewhat photonegative, with a clear diurnal activity cycle with higher activities at nighttime (Strong 1967, Goddard 1979b). It does not tolerate higher temperatures, becoming torpid above 15°C and dying within minutes at 25°C; on the other hand it remains active down to -16°C (Goddard 1979b). *S. villous* apparently feeds on fungal hyphae, algae as well as possibly dead plant material (Gressitt 1967, Strong 1967, Goddard 1979c).

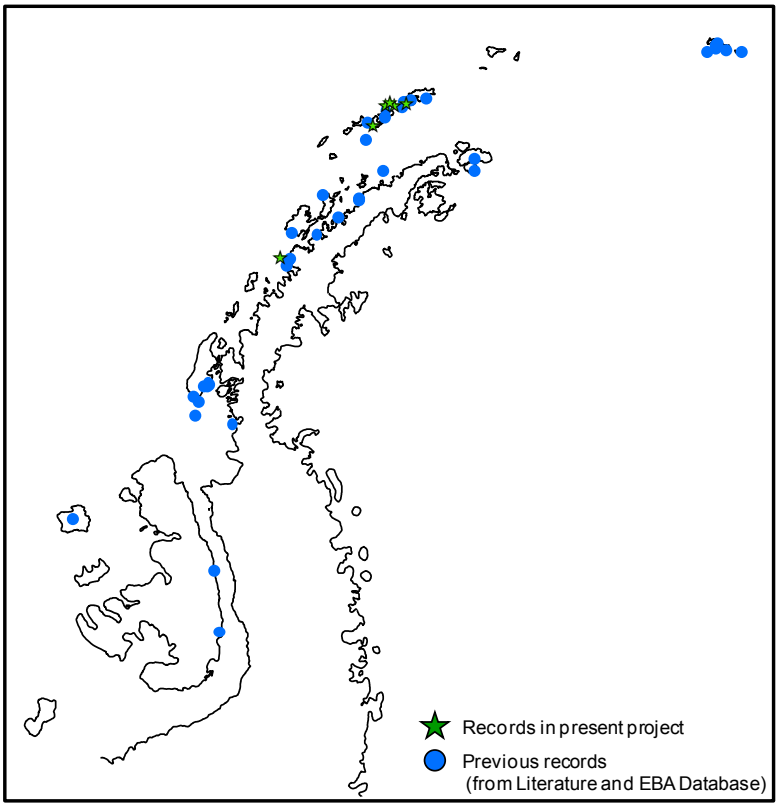


Fig. 108: Previous records of *S. villosus* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

In contrast to its being one of the most frequently found species in previous studies, in the present investigations it was only found in 2010 and usually in only single or few individuals in single samples, albeit in a variety of localities (Fig. 1083). An exception was the second locality at Punta Christian, where the species was found in many samples in larger populations. The species did not correlate significantly to any habitat parameter (Appendix 4, Table A4-2). Its distribution in anthropogenically influenced and non-influenced sites was locality specific (Fig. 109) and, therefore, no overall statistical difference could be ascertained (Appendix 5, Table A5-5).

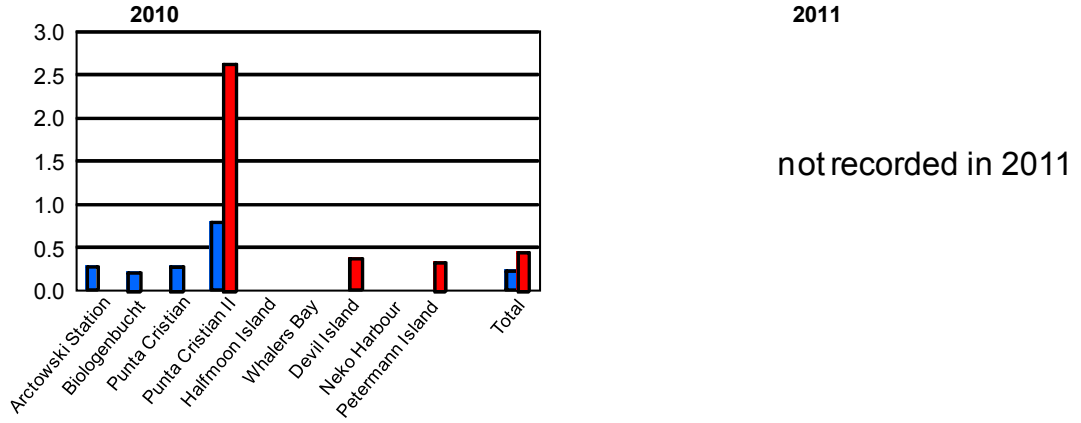


Fig. 109: Total densities *S. villosus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010.

Rhagidia gerlachei (Trouessart, 1903)

Another species commonly found studies of the Antarctic mite fauna is *R. gerlachei*, where it is apparently very widely distributed throughout the maritime Antarctic (Fig. 110). One of two *Rhagidia* species occurring in the Antarctic, it is a large and very active predator feeding mainly on Collembola (Lister 1984 [zit in Convey & Quitana 1997], Gressitt 1967, Strong 1967). It occurs in a wide range of habitats, such as *Prasiola* mats, lichens and mosses (Gressitt 1967, Convey & Quintana 1997), but is most frequently found and in its highest abundances under or on stones (Dalenius 1965, Strong 1967, Richard et al. 1994, Convey & Quintana 1997, Convey & Smith 1997). It apparently requires very high humidity, which in the maritime Antarctic is generally found under stones and rocks (Strong 1967).

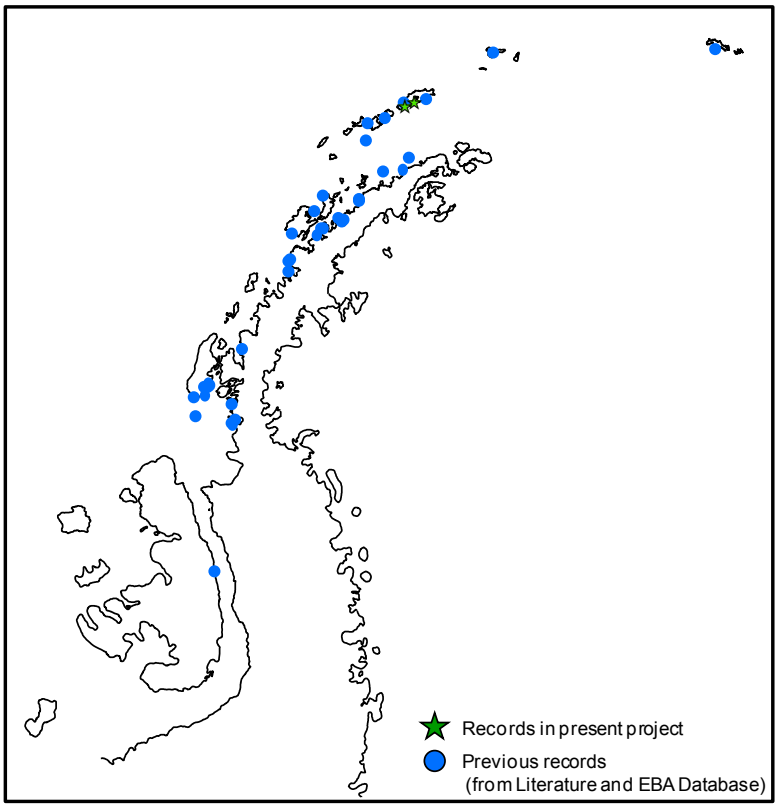


Fig. 110: Previous ecords of *R. gerlachei* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

In the present study, *Rhagidia* species were only found in two sites (in 2010) as a single individual (Fig. 110). That this taxon was not registered in the present study, although it was very frequently found in previous investigations, is most likely due to differences in the substrate sampled (here more soil and vegetation was sampled and not under or on stones and rocks). It is mentioned and discussed here solely because it is such a commonly found species on and around the Antarctic Peninsula. Due to only two individuals having been found no statistical analysis of relationships to habitat parameters or of human influence on its distribution could be carried out.

Ereynetes macquariensis Fain, 1962

Like the species described above, *E. macquariensis* also belongs to the typical maritime Antarctic fauna, whereby it has mostly been found in the northern Antarctic Peninsula, South

Shetland Islands as well as the South Sandwich Islands (Fig. 111). The species is also known from various Subantarctic islands (Pugh 1993, Marshall et al. 1999). In the maritime Antarctic it has been commonly found in algae-rich soils, *Prasiola* mats, mosses, or swards of *Deschampsia antarctica*, whereby the species apparently is most frequent and has its highest densities in mosses (Strandtmann & Tilbrook 1968, Goddard 1979b, Usher & Edwards 1984). In moss turf *E. macquariensis* is often evenly distributed throughout the vertical profile, whereby adults can be found deeper in the profile than juveniles (Goddard 1979a, Usher & Booth 1984). As opposed to other species, this species apparently does not often form aggregations (Usher & Booth 1984). It seems to be particularly susceptible to desiccation (Goddard 1979b), which may explain its occurrence in deeper levels of the soil profile. Eggs apparently hatch to larvae in summer (December and January), whereby other life-cycle stages can be found throughout the year; the species apparently overwinters as last-stage nymphs (Tritonymphs) or adults (Usher & Booth 1986). *E. macquariensis* seems to feed on algae and fungi (Goddard 1979c), but may also be predatory (Usher & Booth 1984).

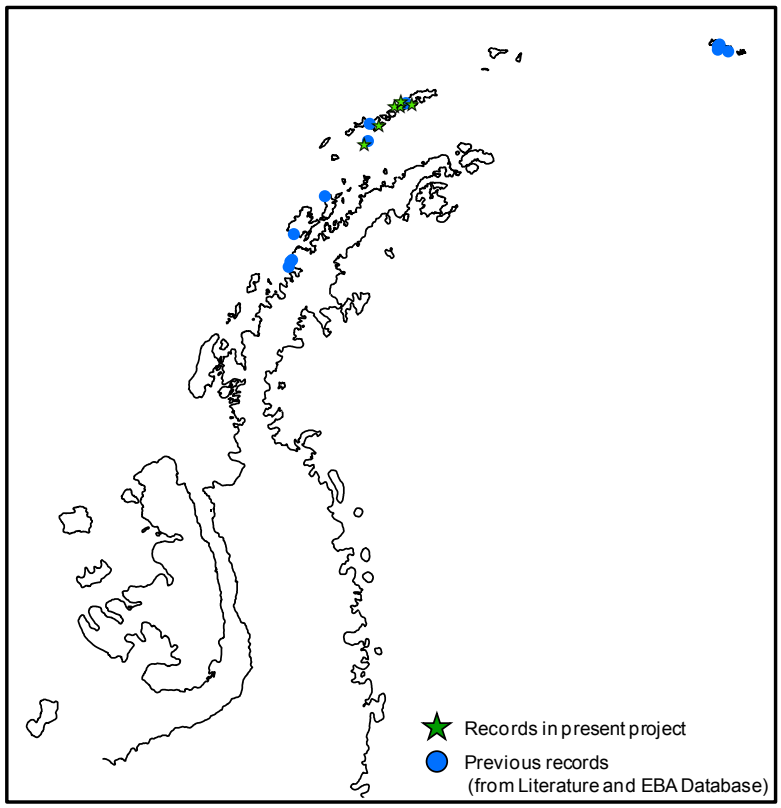


Fig. 111: Previous records of *E. macquariensis* in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

In the present study the species was found primarily in the South Shetland Islands in both study years (Fig. 111), often in numerous individuals spread throughout many samples in those localities where it occurred. Mostly juveniles were registered. *E. macquariensis* correlated positively to vegetational cover in both study years as well as to amounts and quality of organic material in 2010 and to soil moisture in 2011 (Appendix 4, Table A4-2). It correlated negatively to (= lower densities in) finer grained soil substrates in 2010. This species was generally more abundant in areas *not* influenced by humans (Fig. 112), although this was statistically significant only in 2011 or when both years were analyzed together (Appendix 5, Table A5-5).

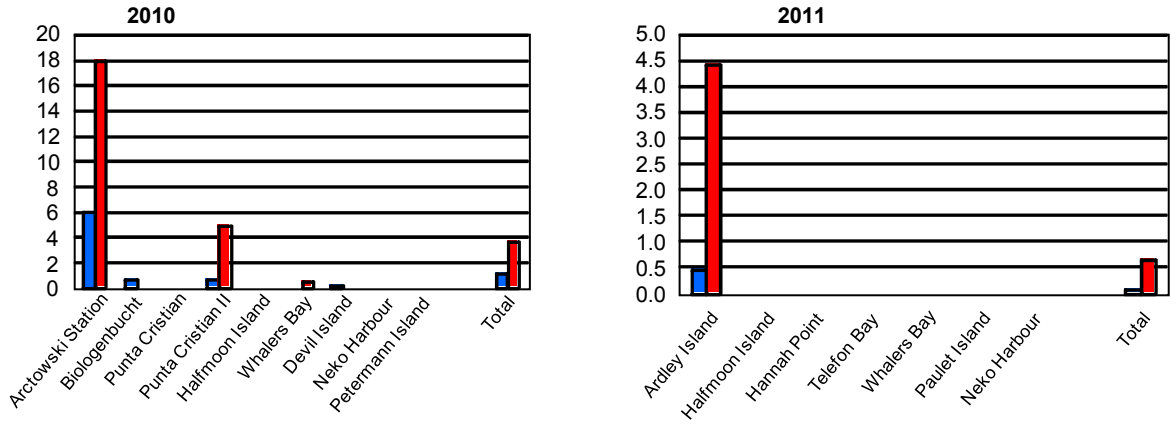


Fig. 112: Total densities *E. macquariensis* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The results of the covariance analysis of the densities of *E. macquariensis* were similar to the correlation and variance analyses. Filtering out habitat parameters revealed in 2010 significantly higher densities in samples with higher vegetational cover (Fig. 113; Appendix 6, Table A6-2) and could show in 2011 a significant statistical interaction between human influence and vegetational cover, with higher densities in anthropogenically influenced areas that had higher vegetational cover (Fig. 114).

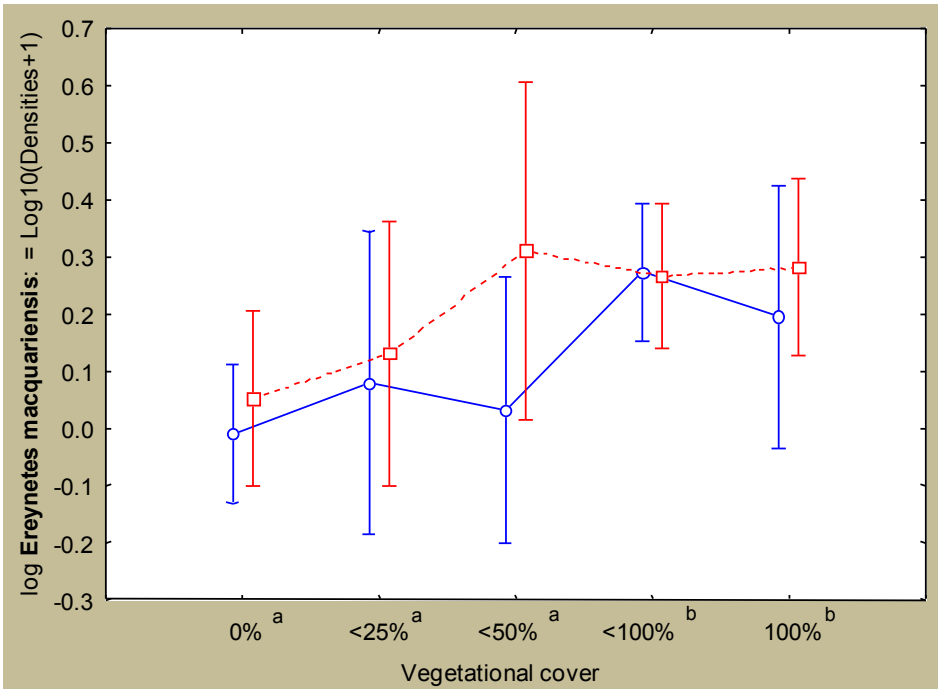


Fig. 113: Results of the covariance analysis (ANCOVA) of the densities of *E. macquariensis* recorded in 2010. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

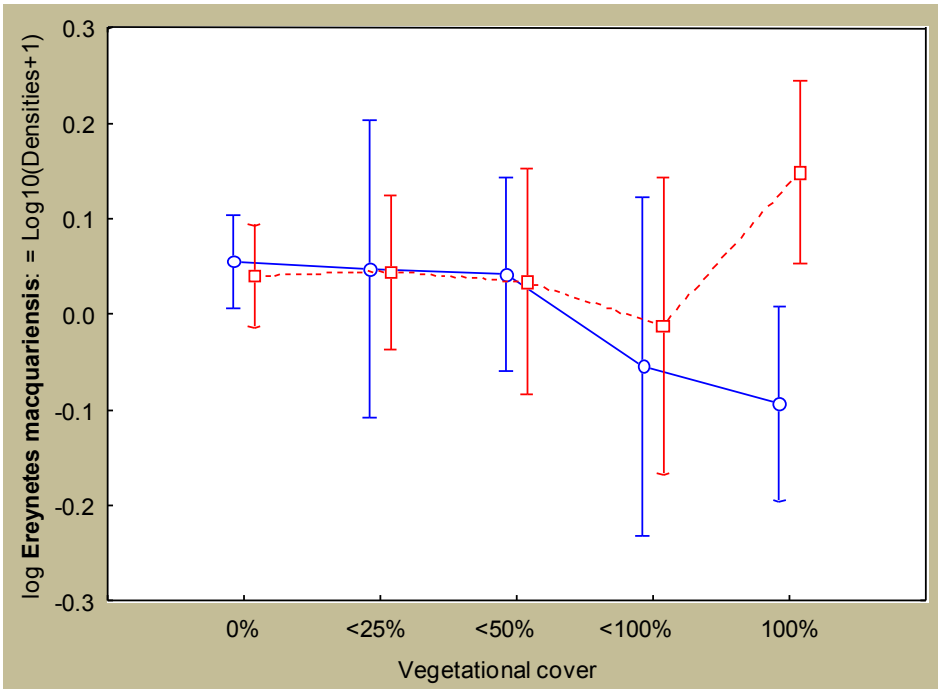


Fig. 114: Results of the covariance analysis (ANCOVA) of the densities of *E. macquariensis* recorded in 2011. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

Apotriophtydeus species

Apotriophtydeus species are very small tydeid mites (= species of the families Meyerellidae, Iolinidae and Tydaeidae), of which five species are known since the taxonomic revision of Usher & Edwards (1986a). Although these species have been found throughout the maritime Antarctic (Fig. 115), little information is given about its habitat preferences. They have been recorded concentrated at 3-6 cm depth within stands of *Deschampsia antarctica* (Usher & Edwards 1984). Usher & Edwards (1986a) described the species as occurring mainly in lichens and mosses, but also described a spatial niche separation of different maritime Antarctic species: *A. penola* in moss-dominated sites; *A. terror* in fellfields, and *A. scotia* in a variety of habitats, the most frequently being drier lichens-dominated habitats and fellfields.

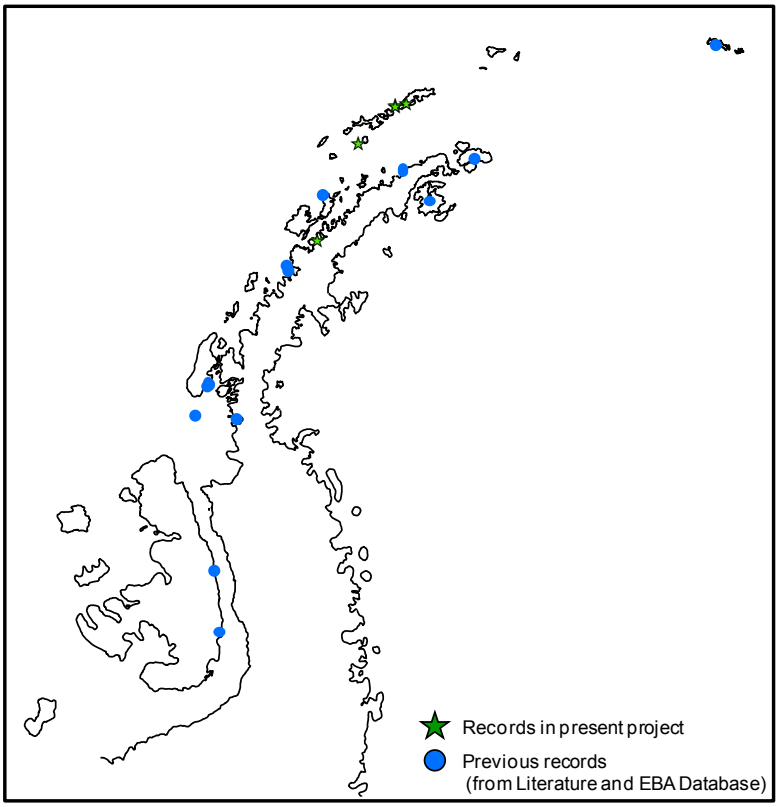


Fig. 115: Previous records of *Apotriophtydeus*-species in the maritime Antarctic (blue dots) as well as records from the current project (green stars). Sources: as in Fig. 94.

The various species of this genus are often taxonomically difficult to differentiate, species separation at times being only possible via a regression of the sizes of different morphological characters. In the present study, it was thus not possible to differentiate all individuals, usually a representative portion of the species of each sample were thus determined. Possibly two species were recognized: *A. scotia* and possibly also *A. wilkesi*. These specimens were primarily found the South Shetland Islands, but also in Neko Harbour (Fig. 115). These specimens were usually registered as one to few individuals in single samples of the various locations, with the exception of Punta Christian, where the species were more widespread and aggregations of very many individuals (> 100) were found in single samples. *A. scotia* only correlated to soil temperature, and this negatively and only in 2010 (Appendix 4, Table A4-2).

Although, in those localities where *Apotriophtydeus* species were found, they were often more abundant in anthropogenically non-influenced areas (Fig. 116), due to the high variability among the separate samples (with very many samples containing no individuals) these quantitative differences were statistically not significant.

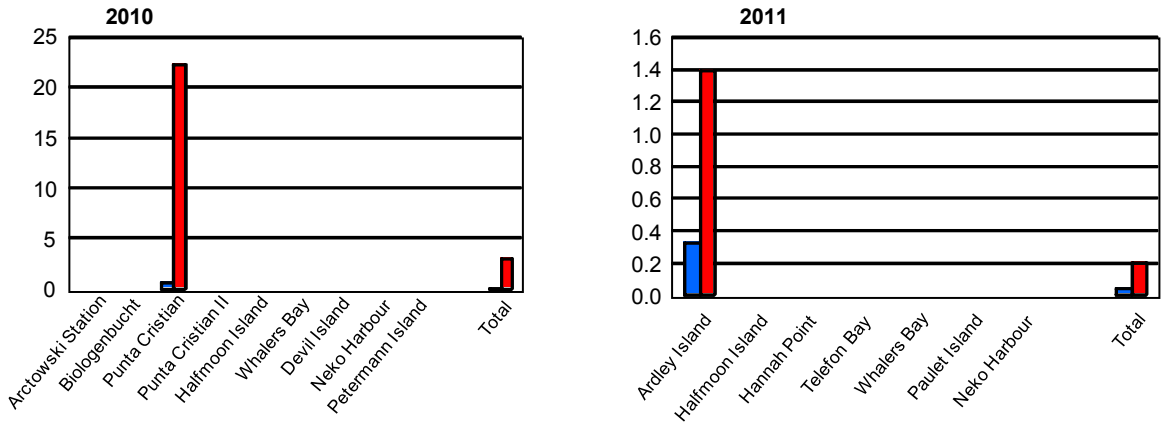


Fig. 116: Total densities *Apotriophtydeus* species (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analysis, on the other hand, could show a significant human influence in 2010, with significantly higher densities in *non*-influenced areas (Appendix 6, Table A6-2). However, this was only true in areas with middle levels of vegetational cover, as revealed in the significant human influence x vegetational cover interaction (Fig. 117).

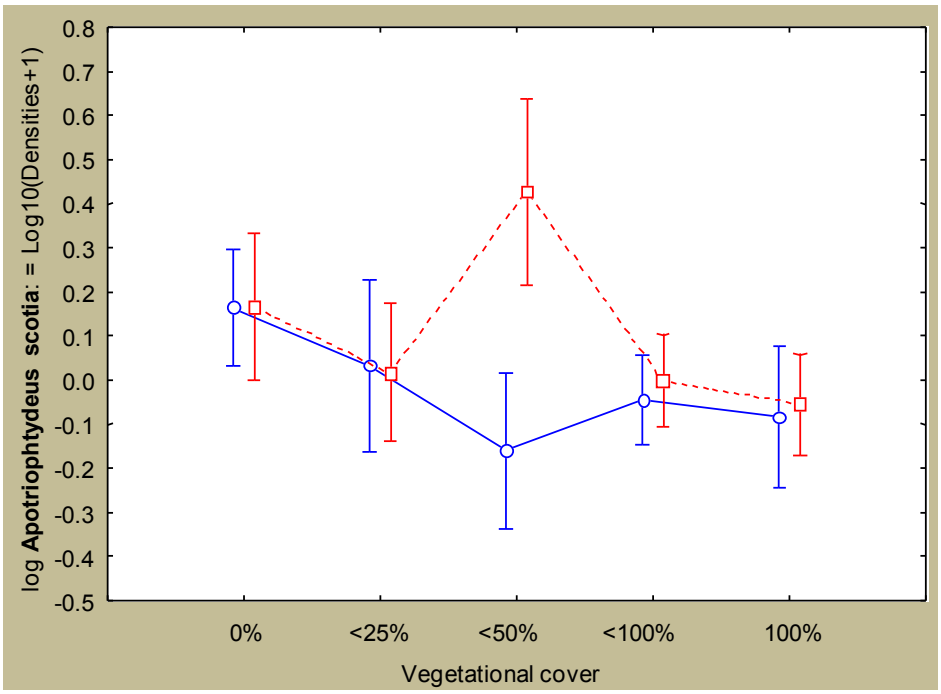


Fig. 117: Results of the covariance analysis (ANCOVA) of the densities of *Apotriophtydeus scotia* recorded in 2010. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

Pretriophtydeus tilbrooki (Strandtmann, 1967)

P. tilbrooki represents another tydeid mite that has been very frequently registered throughout the maritime Antarctic in very many studies (Fig. 118). As in *Apotriophtydeus* species, this species also represents one of the smallest Antarctic mites (Goddard 1979b). It has been registered in various habitat types, whereby the various studies are often contradictory in this regard; e.g., it has been found both under stones and in vegetation (Strong 1967), was

considered scarce in moss turf of Signey Island (Usher & Booth 1984), was found only in mosses and not under stones or in algal mats (Convey & Quintana 1997), but then again in large aggregations in *Prasiola* and lichens with only few specimens found in mosses (Goddard 1979b). Thus, other factors other than the vegetational cover seem to determine the species' occurrences. *P. tilbrooki* seems to be less prone to desiccation than other Actinedid species and also shows no photonegative behavior (Goddard 1979b). The adult seems to be the main overwintering stage (Goddard 1979a), as opposed to many other mite species, which (also) overwinter as juveniles. This species seems to feed predominantly on algae, fungal hyphae and lichens (Goddard 1979c, Strong 1967), but may also be predatory (Convey et al. 2000a).

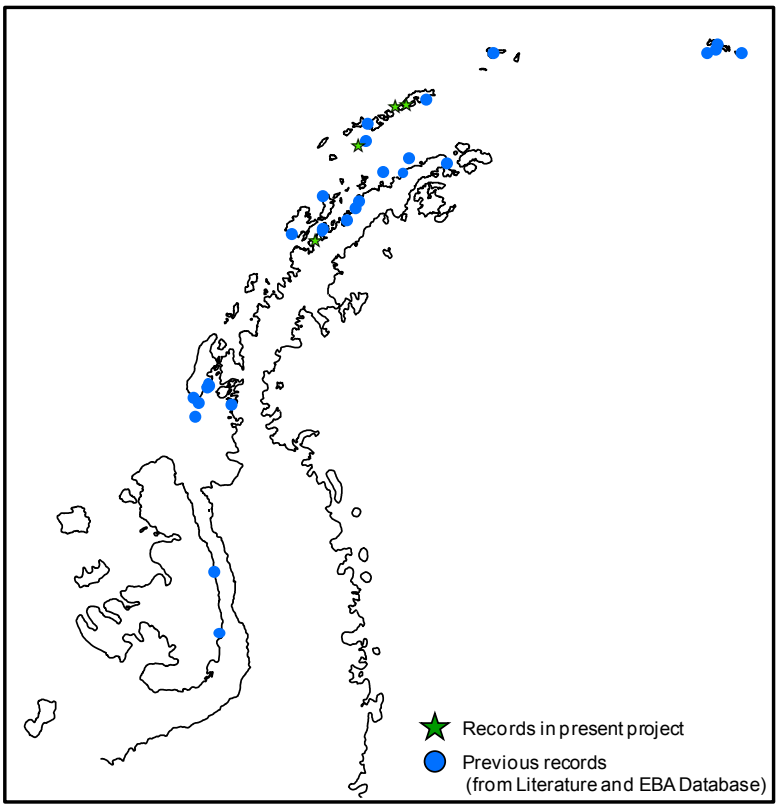


Fig. 118: Previous records of *P. tilbrooki* in the maritime Antarctic (blue dots) and records from the current study (green stars). Sources: as in Fig. 94.

In the present study *P. tilbrooki* was only found in a few localities (Fig. 118) and even there only as single individuals in sporadic samples. This is in sharp contrast to its wide distribution reported in the literature. Due to so few individuals being registered, it was not possible to statistically analyze its relationship to habitat parameters or the human influence on its distribution and is mentioned here primarily due to its otherwise frequent occurrence in the maritime Antarctic.

Individuals of the related family Tydeidae were also recorded in the present study. Only a few individuals were found as single specimen in sporadic samples spread throughout all the studied localities. Based on the possible determination (only possible in adults, of which there were very few), these specimens have been tentatively labeled "*Lorryia*", although it is likely that more than one genus is included here. Dozens of species of this genus exist worldwide (Kazmierski 1998), the taxonomy of which is extremely difficult, for which reason they cannot be evaluated more closely here. Species of this genus have been listed for maritime Antarctic

localities, but were species parasitic on seals and were considered to be probably dislodged from the host animals (Pugh 1993), which is also quite possible in the present study.

Bakerdania cf. *antarcticus* (Mahunka, 1967)

Bakerdania antarcticus is a small Pygmeoid mite first described in 1967 from the Antarctic Peninsula (Danco Coast near the Chilean Base Gabriel Gonzalez Vedela; Makunka 1967). The determination in the present samples is somewhat uncertain, because the original description is incomplete and this genus is very species rich worldwide, so that a potential misdetermination is possible. However, this is the only species of this genus having been recorded in the Antarctic, although other species are known from the Subantarctic (i.e., Cross 1964, 1970). Up to now this taxon has only been recorded from the South Sandwich Islands, Livingston Island and Deception Island (Fig. 119). Little is known about its habitat preferences, with its only mention being having been in association with birds' nests (Tilbrook 1967b, Goddard 1979b *as Pygmephorus* sp.).

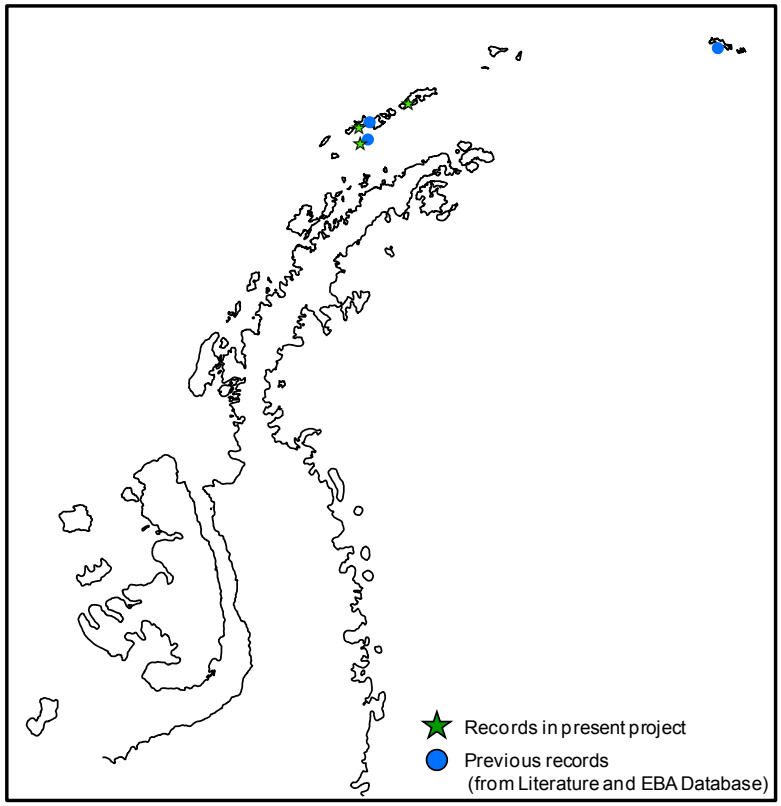


Fig. 119: Previous records of *B. antarcticus* in the maritime Antarctic (blue dots) and records from the current project (green stars). Sources: as in Fig. 94.

In the present study, *B. antarcticus* was only found in three locations of the South Shetland Islands (Fig. 119). Although sometimes only occurring in few individuals in single samples within a locality, the species was actually fairly abundant and widespread near Arctowski Station (King George Island) as well as in Whalers Bay (Deception Island). The species correlated in 2010 positively to location, but also to soil moisture (Appendix 4, Table A4-2).

Although in 2010 near Arctowski Station actually a few more individuals were registered in the anthropogenically non-influenced areas, the samples of these were larger than those of the influenced areas. Therefore, when the results of the species were transformed into densities per

volume, the densities were higher in the influenced areas (Fig. 120). In Whalers Bay of this year, the species was only found in non-influenced areas. Thus, overall for 2010 the differences in the species densities between influenced and non-influenced areas were statistically not significant. On the other hand, in the year 2011 *B. antarctica* was primarily found in non-influenced areas, so that its abundances were also significantly higher in these areas (Appendix 5, Table A5-5). When both years were analyzed together no statistically significant human influence could be determined.

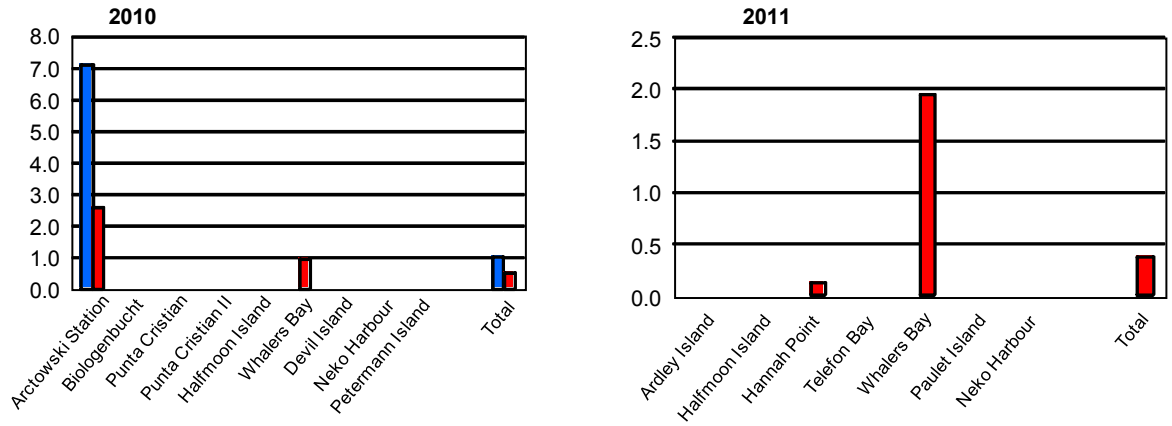


Fig. 120: Total densities *B. antarcticus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analyses reflected and confirmed the results of the variance analyses. In the year 2010 these revealed that *B. antarcticus* was present in significantly higher densities the higher the vegetational cover was (Fig. 121; Appendix 6, Table A6-2). In the year 2011, these analyses could show a interaction between human influence and vegetational cover (Fig. 122), with higher densities in anthropogenically *non*-influenced areas with high vegetational cover.

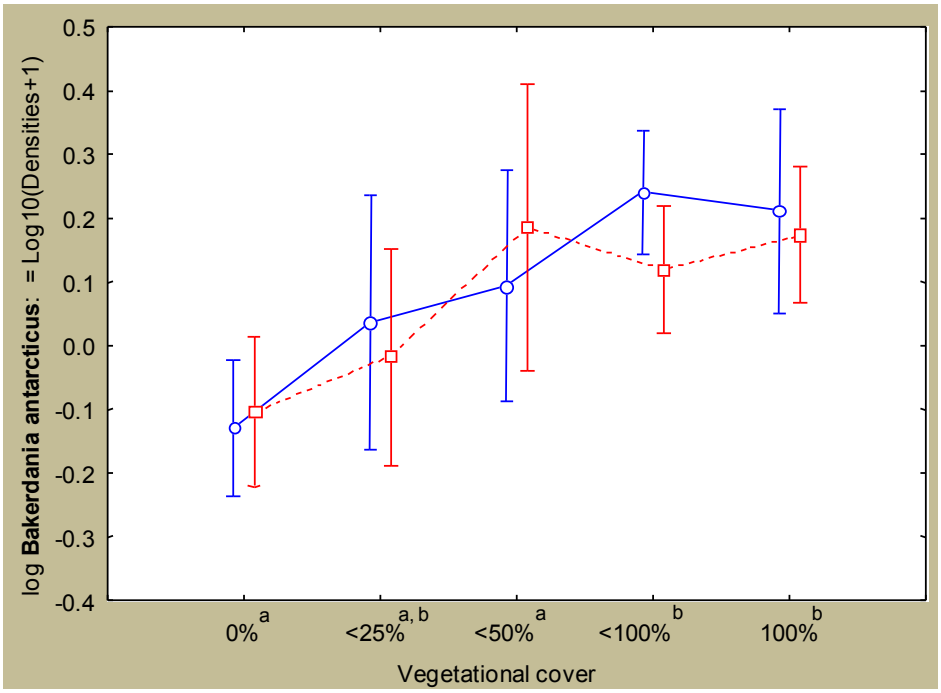


Fig. 121: Results of the covariance analysis (ANCOVA) of the densities of *B. antarcticus* recorded in 2010. Figure explanation as in Fig. 89. Negative values

(y-axis) are statistical artefacts and result from the logarithm of very low densities.

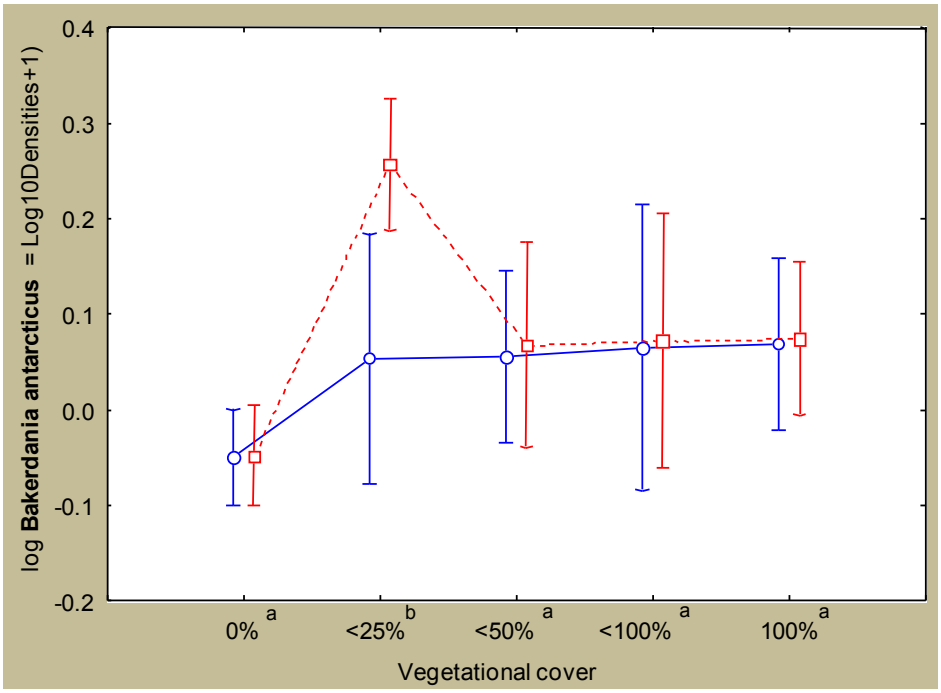


Fig. 122: Results of the covariance analysis (ANCOVA) of the densities of *B. antarcticus* recorded in 2011. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

A number of other individuals from families related to this taxon were also registered, particularly Tarsonimidae taxa. Although found throughout very many localities in the present study (Appendix 3, Table A3-2), these were usually only registered as single individuals per sample and were usually juveniles. Their taxonomy is extremely difficult, so that a number of adult individuals are necessary for a secure determination, which was thus not possible here. Although taxa of this family have only been listed as occurring in the Subantarctic by Pugh (1994), based on the present data and level of determination they cannot be evaluated whether they are native to the Antarctic or introduced. Species from this family are often algivorous, fungivorous as well as phytophagous (Krantz & Walter 2009), feeding preferences that coincide with many of the species determined in this study.

3.4.2.1.2.2 Potentially introduced (non-native) species

Besides the taxa described above, a number of genera and species were determined in the present studies that to date have not been recorded anywhere in the Antarctic before. Due to their known distribution, they can be considered to be at least potentially introduced into the maritime Antarctic, although this cannot be proven with absolute certainty.

Alicorhagia Berlese, 1910 spec

Only a single individual of this taxon was found on Whalers Bay (Deception Island) in 2010. Although only one individual was identified in the present study, its registration here is remarkable since, on the one hand, the taxon has never before been proven in Antarctica and, on the other hand, its morphology is conspicuous, so that is very difficult to overlook. Only ca. five species are known from this genus. However, no determination keys exist for the genus

and, without all original descriptions, it was not possible to identify this individual to species level. The genus occurs worldwide in many various habitats, from deserts to forests, so that the origin of this taxon may possibly lie elsewhere than the maritime Antarctic. This parthenogenetic genus is omnivorous, feeding on algae and fungi but also requiring nematodes in its diet (Kethley 1990). Since only one individual was found, the taxon occurs in, at most, very low population densities. The possibility cannot be excluded that this specimen represents only a sporadic, recently introduced individual that could not establish a viable population.

cf. *Coccotydaeolus krantzii* Baker, 1965

Coccotydeolus is a very small tydeid genus for which little is known about its distribution, although it has been found repeatedly worldwide in xerothermous sand habitats (Estrada et al. 1988; Cepado-Pizaaro and Whitford 1989; Sanchez-Rocha and Palacios-Vargas 1996; Russell & Alberti 2009). The genus is rarely identified to species level, although the individuals determined here fit the description of *C. krantzii* very well with one exception: a single aspect of its chaetotaxy (number and position of hairs; here particularly on the first segment of leg one) does not correspond to the morphology of the genus. Thus the determination must still be considered tentative; however, to erect a new taxon based on the lack of one character does not seem reasonable. Although this species was only found in single individuals in 2010, its record in the present study is remarkable due to very many specimens found in 2011 and its overall distribution was fairly widespread throughout the sites studied in these investigations (Fig. 123), being particularly abundant in Whalers Bay (Deception Island) and Neko Harbour. It is thus apparent that this taxon exists in viable populations in the maritime Antarctic.

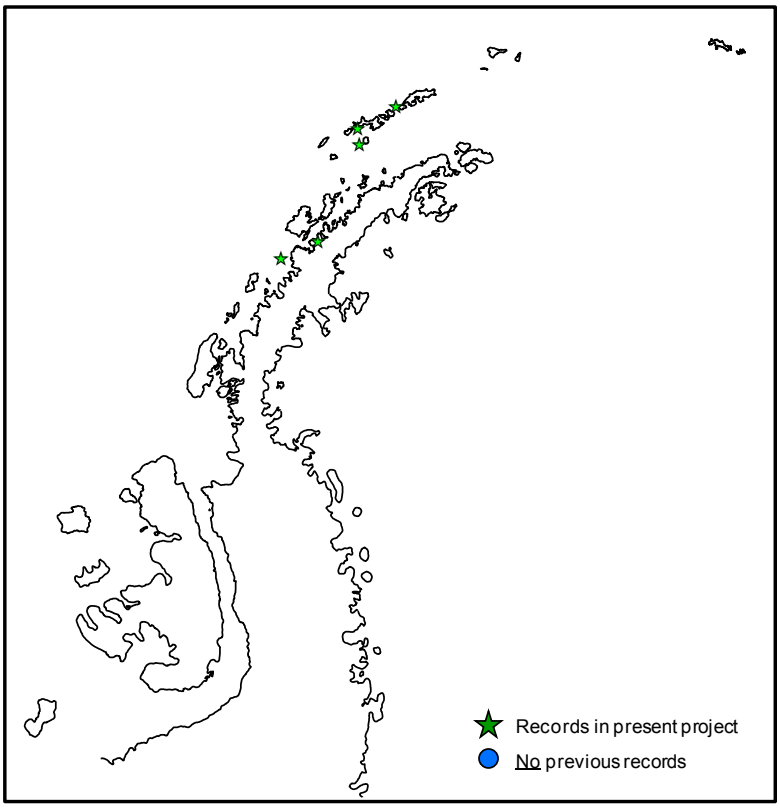


Fig. 123: Records of *C. krantzii* in the maritime Antarctic from the present project (green stars).

Although superficially similar to other small tydeid species occurring in the Antarctic (i.e., *Paratydeolus*; Usher & Edwards 1986c), the morphological characters commonly used for their determination are considerably different. Therefore, it is highly unlikely that these taxa were incorrectly determined in the past or in the present study. It is also therefore doubtful that the species was overlooked in the past. This is remarkable considering its widespread distribution found in the present study. On the other hand, this species was found in the present project – with the exception of Whalers Bay - in sites that have not been previously studied acarologically.

In 2011, the study year in which the most individuals were found, a negative correlation to amounts of N and C (signifying nutrient-poor soil substrates) could be discerned (Appendix 4, Table A4-2). Although in the present study *C. krantzii* was found in higher abundances in the anthropogenically influenced areas (Fig. 124), this result was statistically not significant.

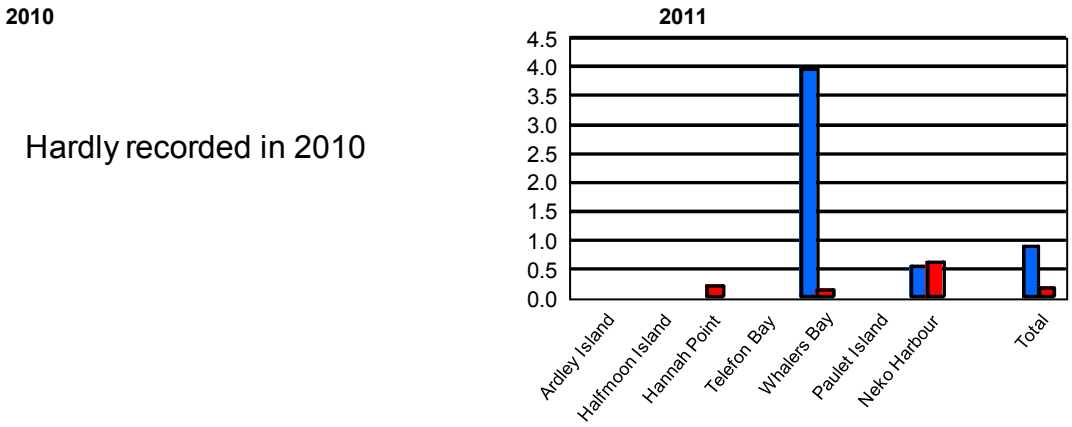


Fig. 124: Total densities *C. krantzii* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in 2011.

Filtering out various habitat parameters, the covariance analysis revealed a significant relationship between vegetational cover and the densities of *C. krantzii* (Fig. 125; Appendix 6, Table A6-2), with this species occurring in its highest densities in samples without any vegetation.

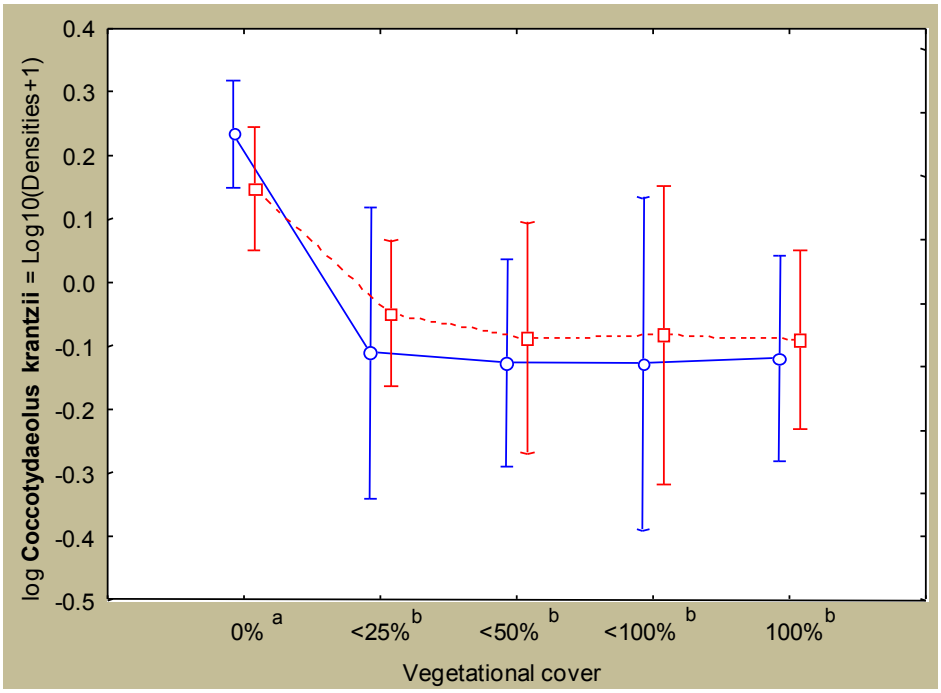


Fig. 125: Results of the covariance analysis (ANCOVA) of the densities of *C. krantzii* recorded in 2011. Figure explanation as in Fig. 89. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

Speleorchestes Trägårdh, 1909 sp.

Speleorchestes sp. is another taxon found widespread in the present study sites (Fig. 126), although almost always as single individuals. Although Pugh (1993) lists the genus as having been found once in the continental Antarctic, this appears to be a misinterpretation of the cited paper. This paper (Rouseville & Greenslade 1988) is a morphological comparison of this genus and *Nanorchestes*, where only the latter genus originated from continental Antarctica. These authors actually list ecological differences between the two genera, stating that *Nanorchestes* occurs in colder and moisture habitats while *Speleorchestes* is normally found in hot and dry habitats. Thus, this taxon has not been found to date in the Antarctic. The genus occurs worldwide, often in dryer sandy habitats (Wallwork 1972, Franco et al. 1979, Steinberger et al. 1990, Cepeda-Pizzaro et al. 1996, Noble et al. 1996, Russell & Alberti 2009, Elmer et al. 2010). Although very small, it is morphologically very distinct and thus unlikely overlooked in previous studies.

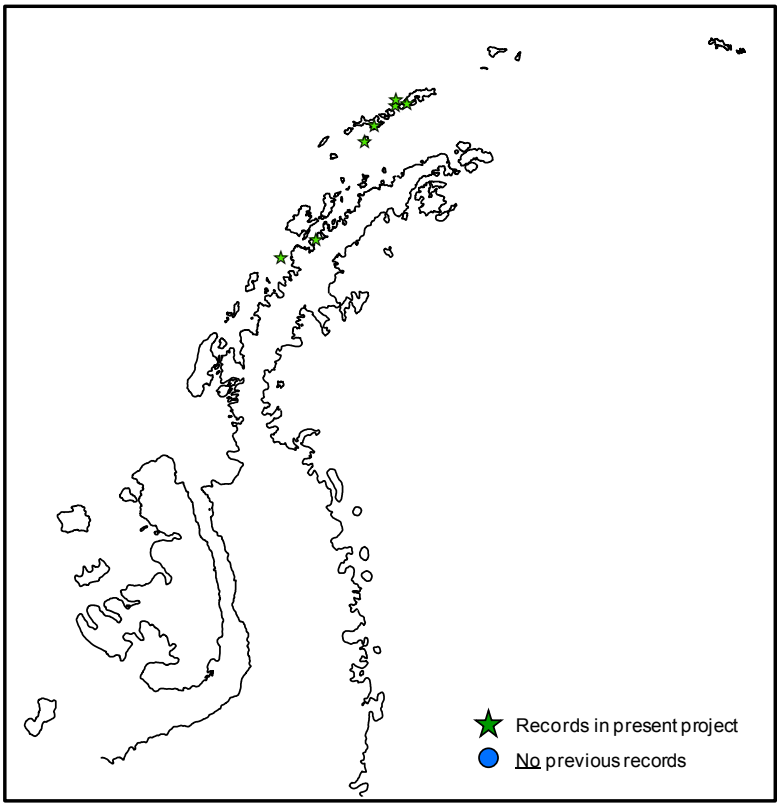


Fig. 126: Records of *Speleorchestes* sp. in the maritime Antarctic from the present project (green stars).

As in the previous species, it is thus remarkable that it was found in many of the localities studied in the present investigation, where viable populations of this taxon can be considered to exist. Also as in the previous species, this genus was found in the present project mostly in sites that have not been previously studied acarologically. Due to its widespread distribution throughout the world and the possibility of it having been overlooked in the past being highly unlikely, this taxon's origin is most likely not in the maritime Antarctic and is probably non-native. Although this taxon was also more often found in human influenced areas (Fig. 127), human effects on its abundances were statistically not significant (Appendix 5, Table A5-5).

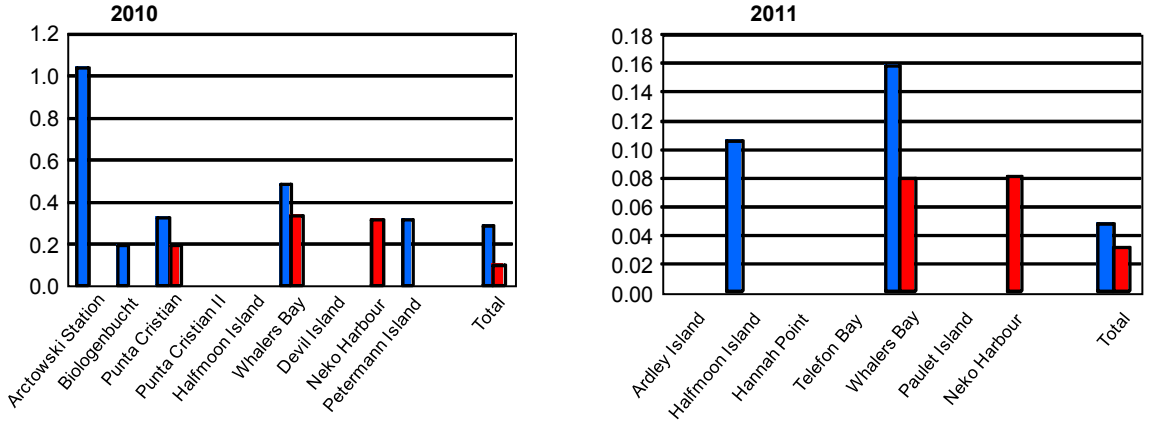


Fig. 127: Total densities *Speleorchestes* sp. (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Terpnacarus gibbosus (Womersley, 1944)

T. gibbosus is a fairly large, primitive mite with a very distinct morphology. The present study is its first record in any location of the Antarctic. It is highly unlikely that it has been overlooked in the past, so that is most likely non-native to the Antarctic. The species has been found on various continents around the world, particularly but not only in the southern hemisphere (e.g., Theron 1976, McDaniel & Theron 1979, Walter 2001), albeit under various names. These different species have been synonymized (Walter 2001), so that the species truly has a worldwide distribution. In the present study it was found only as single individuals from King George Island as well as Whalers Bay (Deception Island) and Neko Harbour (Fig. 128). Due to only eight specimens in total having been registered, an analysis of potential human influence on the species could not be performed. However, the registration of multiple individuals indicates that the species occurs in viable populations.

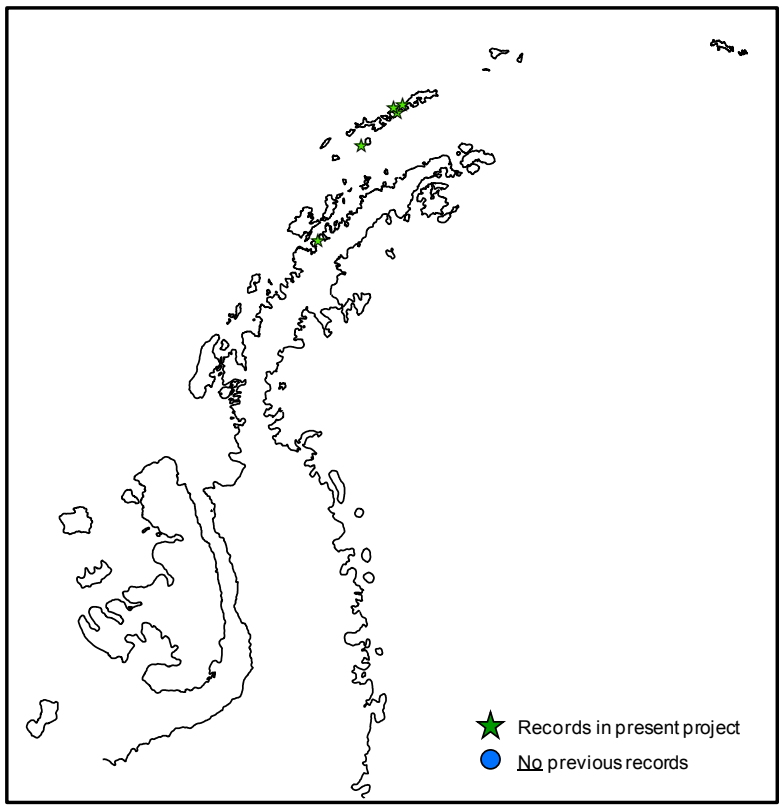


Fig. 128: Records of *T. gibbosus* in the maritime Antarctic from the present project (green stars).

3.4.2.2 Oribatida

3.4.2.2.1 General community parameters

A total of 1107 individuals of the Oribatida were registered in the present project, 938 in the study year 2010 and 169 in 2011. These individuals were registered in total densities between zero in many of the localities and over 100 individuals per 100 cm^3 substrate, e.g., in Halfmoon Island in the year 2010 (Fig. 129). In both study years, significant differences in the total Oribatid densities between the various localities could be determined (Appendix 5, Table A5-1).

Although almost an order of magnitude more individuals were recorded 2010 than in 2011, in those localities studied in both years the differences in densities between the two years were statistically not significant. Even in, for instance, Halfmoon Island, where the densities recorded in 2010 were almost 100 times larger than those of 2011 (Fig. 130), these differences were statistically not significant. This was due to the fact that the higher densities of 2011 were generally caused by most individuals being recorded in a large aggregation of a single sample.

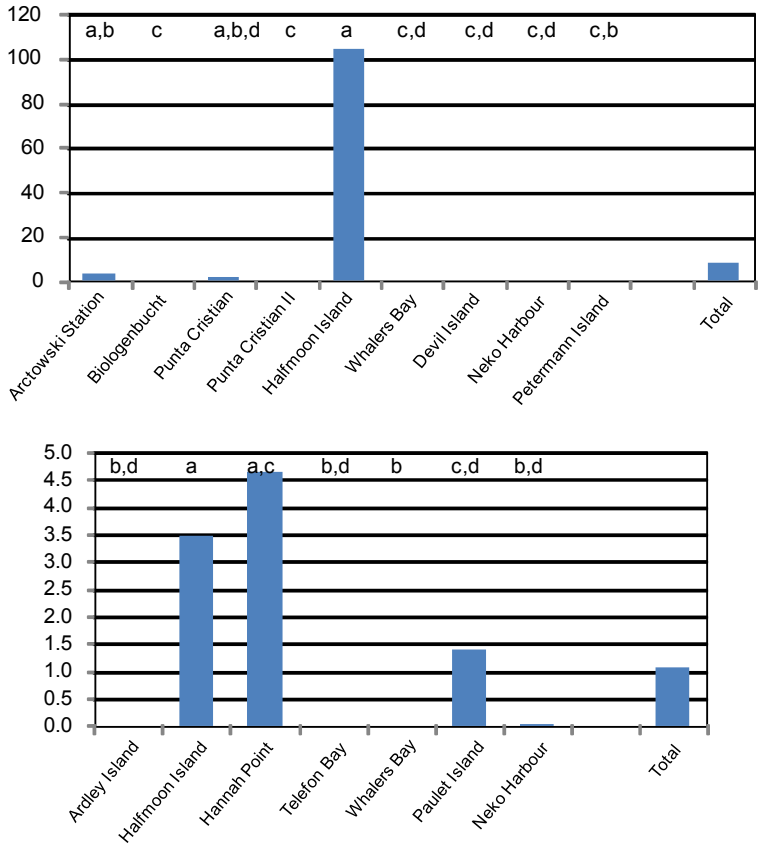


Fig. 129: Average total Oribatid densities (per sample, in individuals per 100 cm^3 substrate) recorded in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= densities in localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

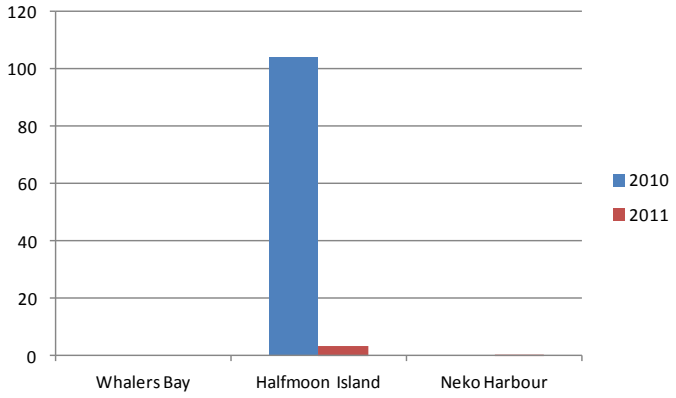


Fig. 130: Total densities of the Oribatida (in individuals 100 cm^{-3}) recorded in 2010 and 2011 in the localities studied in both years.

Only few consistent and interpretable correlations between total densities of the Oribatida and the background habitat parameters could be discerned (Appendix 4, Table A4-3). Truly clear was only a positive relationship in both years between total densities and the parameters that characterize the amounts (organic matter %), composition (contents of C and N, both in %) as well as quality (C/N ratio) of the organic substance of the soil substrates. In 2010 densities also correlated positively to soil moisture and, in 2011, positively to the vegetational cover as well as negatively to soil pH and soil temperature.

Regarding the human influence on the total densities of the Oribatida, the results of the two study years were contradictory. In the year 2010, densities were significantly lower in areas influenced by human trampling (Fig. 131; Appendix 5, Table A5-6). In 2011, on the other hand, densities were higher in the anthropogenically influenced areas, but these results were statistically not significant. When the data from both years were analyzed together, due to the much higher densities in 2011, significantly lower densities could again be discerned in the human influenced areas.

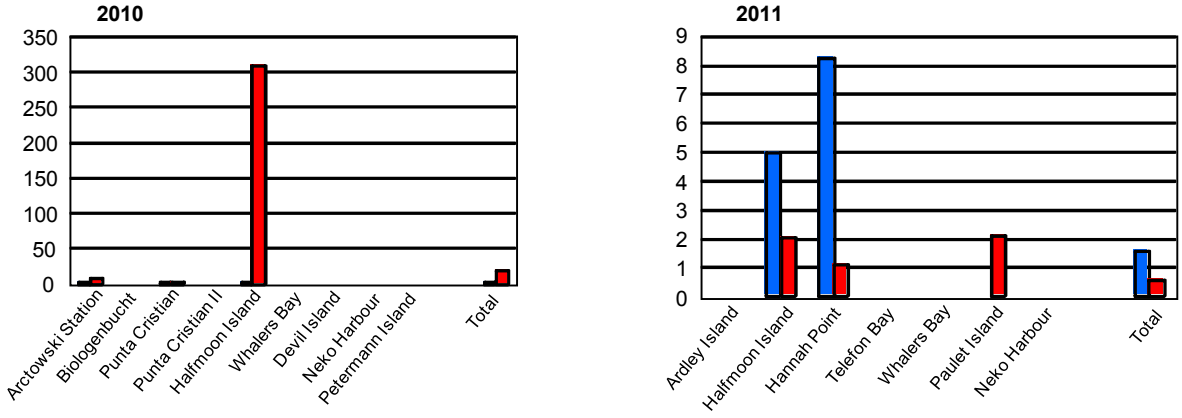


Fig. 131: Total oribatid densities (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

After filtering out the variability caused by various background habitat parameters, the covariance analysis confirmed and made even clearer the conflicting results of the variance analyses concerning the anthropogenic influence on total oribatid densities. In 2010 individual densities were significantly higher in the non-influenced areas (Fig. 132; Appendix 6, Table A6-3). These differences were stronger in samples with vegetation, as signified by the statistically significant interaction between human influence and vegetational cover.

In 2011, however, the opposite was true, where densities were (according to the covariance analysis) significantly higher in the human influenced areas (Fig. 133; Appendix 6, Table A6-3). In this year, total densities of the Oribatida were significantly larger in samples with more vegetational cover. Therefore, the larger densities in human-trampled areas were significantly higher in those areas with a stronger vegetational cover (interaction human influence x vegetational cover).

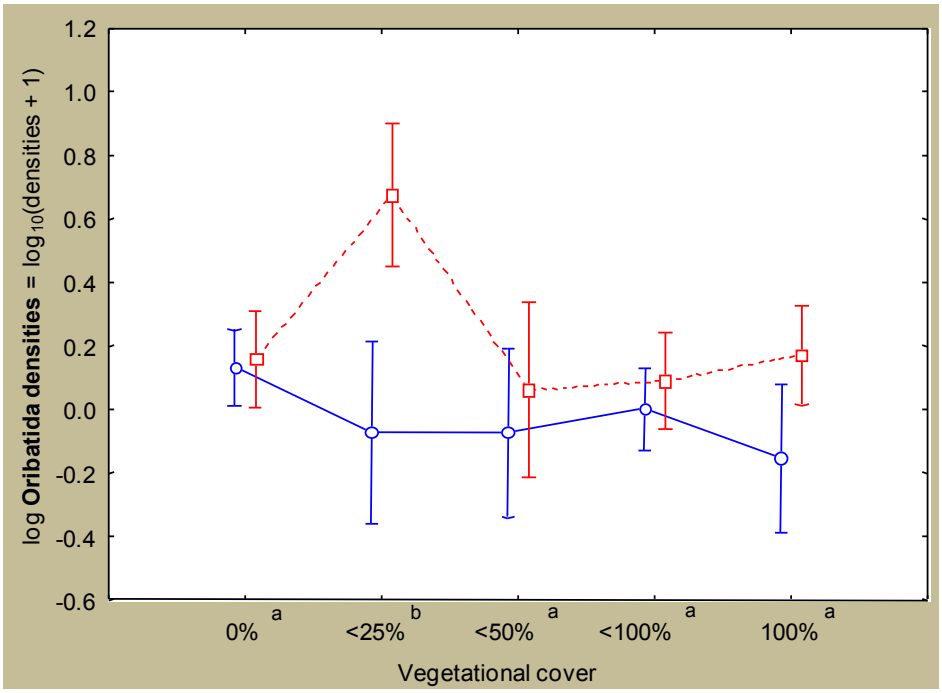


Fig. 132: Results of the covariance analysis (ANCOVA) of total oribatid densities recorded in 2010 after filtering out various background habitat parameters. Densities (given in log individuals per 100 cm^3 substrate) in anthropogenically influenced areas in blue and in non-influenced areas in red. Different letters denote significant oribatid density differences between vegetational cover categories.

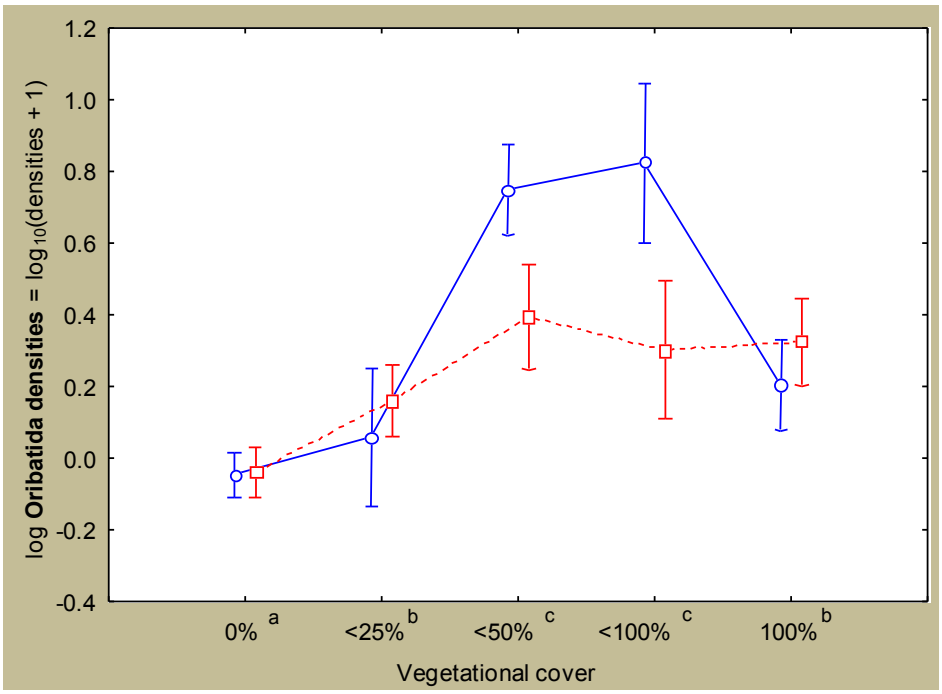


Fig. 133: Results of the covariance analysis (ANCOVA) of total oribatid densities recorded in 2011. Densities in anthropogenically influenced areas in blue and in non-influenced areas in red. Figure explanations as in Fig. 132.

In those localities where Oribatida were recorded, species richness was comparatively low. A total of five species were registered in the two study years, four species in 2010 and only two in 2011. Species richness per locality with Oribatida ranged from a single species to maximum of

three taxa. It must be noted that, in both study years, the vast majority of registered individuals were juveniles (nymphs), which cannot be securely determined to species level.

Strongly significant differences in species richness between the various localities were also discerned (Fig. 134; Appendix 5, Table A5-1). Average species richness in the various localities generally followed the differences in total densities of Oribatida, with the exception of some of the study sites and King George Island, which showed very low densities but higher species richness.

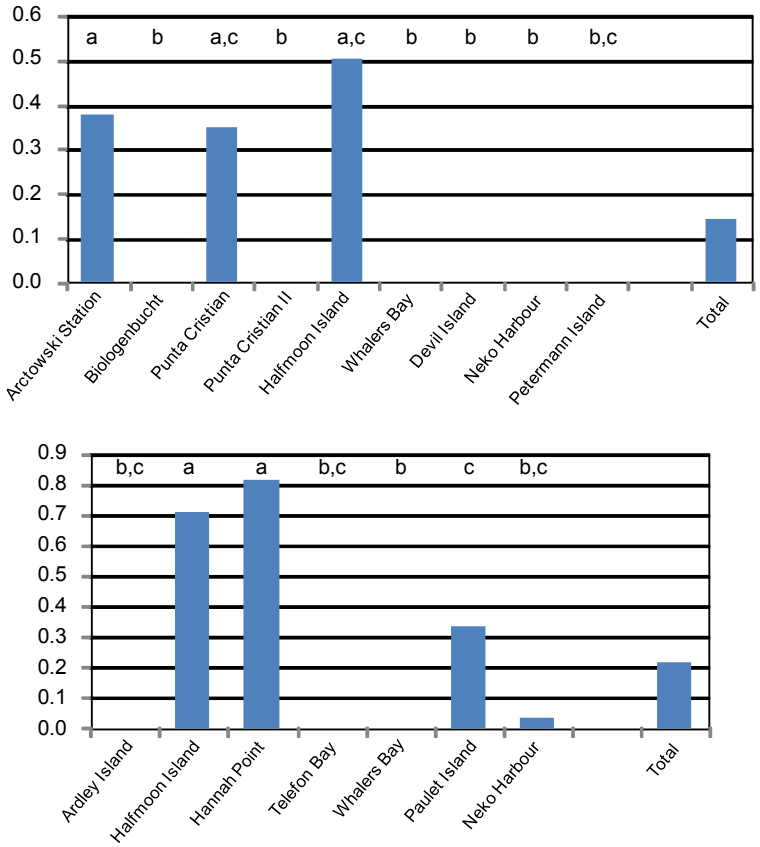


Fig. 134: Total oribatid species richness (in average number of species per area) observed in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

Although in total fewer species were recorded in 2011 than 2010, in those localities studied in both years average species richness (in number of species per sample) was actually somewhat higher in 2011 (Fig. 135) due to more samples containing Oribatida in this year. However, since never more than one species per sample was registered in these localities, these yearly differences were statistically not significant.

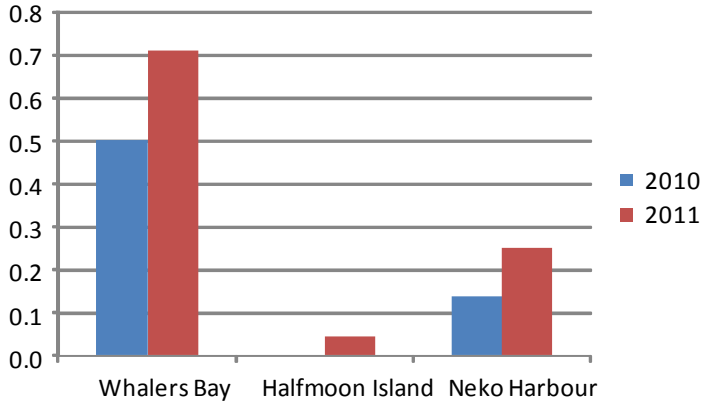


Fig. 135: Average species richness of Oribatida (in average number of species per sample) recorded in those localities sampled in the two study years.

The correlations of species richness with habitat background parameters generally followed those of total oribatid densities (Appendix 4, Table A4-3). In both years, species richness related positively to the amounts of organic material in the soil substrates as well as to vegetational cover. In 2010 the number of registered species related positively to soil moisture and, in 2011, negatively to soil temperature and pH.

Concerning the anthropogenic influence of the sampled areas, species richness was almost always higher in the non-influenced areas (Fig. 136). However, these differences were only statistically significant in 2010 or when both years were analyzed together (Appendix 5, Table A5-6).

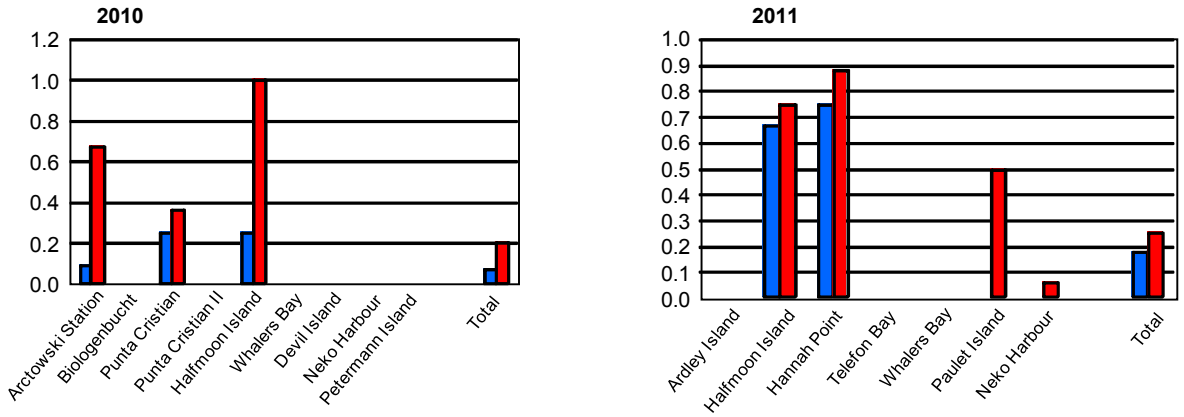


Fig. 136: Total oribatid species richness (in average number of species number per area) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The results of the covariance analyses generally paralleled those of the variance analysis, especially in the year 2010. In this year, according to the ANCOVA, species richness was significantly higher in the anthropogenically non-influenced areas (Fig. 137; Appendix 6, Table A6-3). In the following year 2011, these analyses could mostly show that species richness became significantly larger with increasing vegetational cover (Fig. 138).

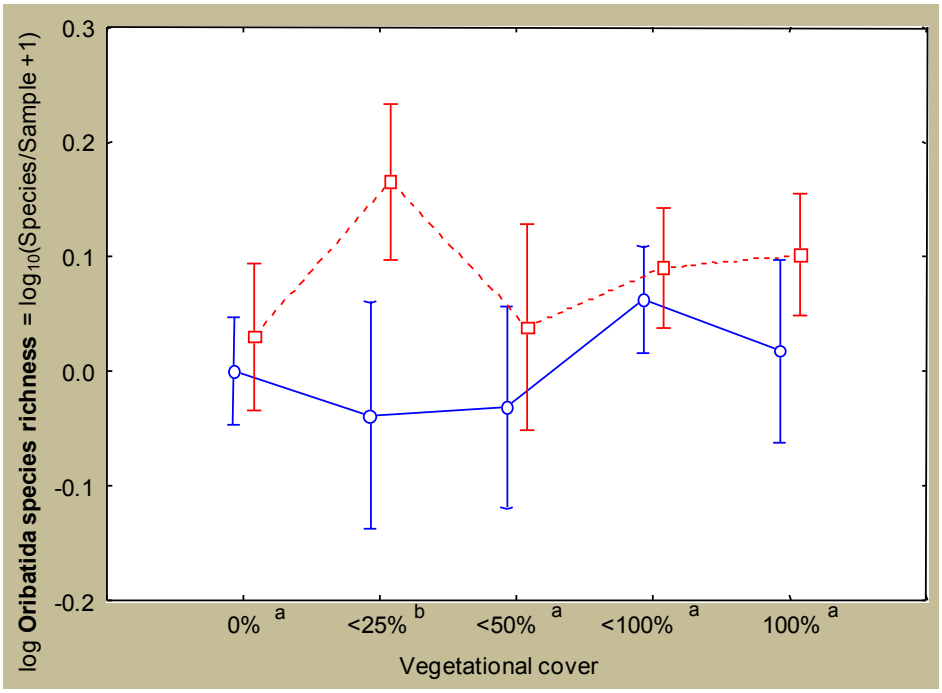


Fig. 137. Results of the covariance analysis (ANCOVA) of average species richness of the Oribatida recorded in 2010 after filtering out various background habitat parameters. Species richnesses in anthropogenically influenced areas in blue and in non-influenced areas in red. Figure explanations as in Fig. 132.

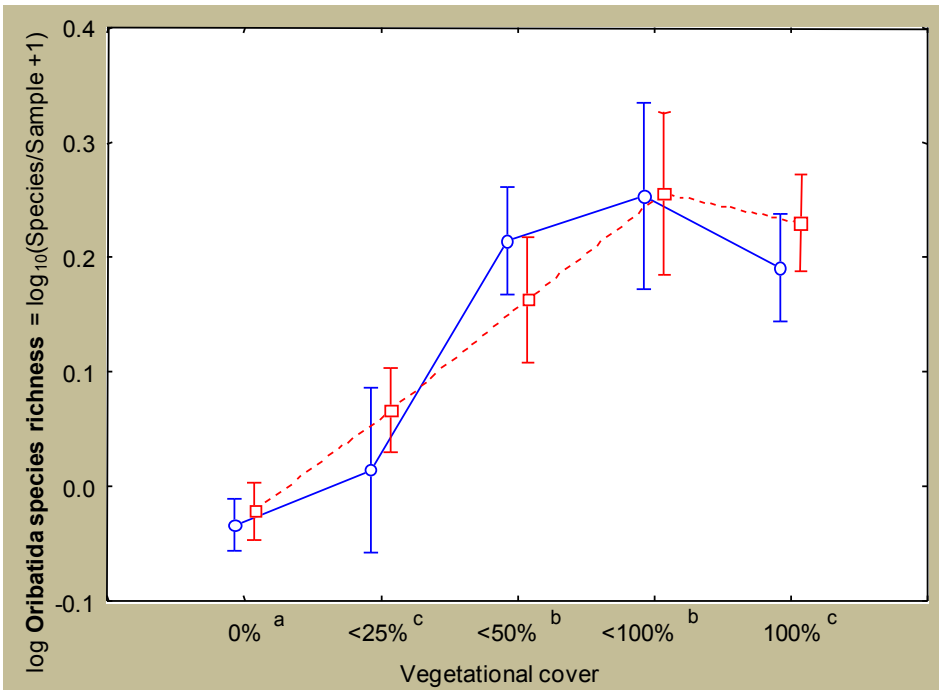


Fig. 138. Results of the covariance analysis (ANCOVA) of average oribatid species richness recorded in 2011. Species richnesses in anthropogenically influenced areas in blue and in non-influenced areas in red. Figure explanations as in Fig. 132.

3.4.2.2.2 Results and descriptions of the determined species

In this study a total of five species of Oribatida were recorded. The literature used for the species determinations are given in Table 9; their total densities in the various study locations

in Appendix 3, Table A3-3. Unfortunately, one of the species (*Brachychochthonius sp.*) could not be determined to species level and a further species (*Liochthonius* cf. *mollis*) could not be assuredly determined, due to the fact that both taxa were only present in few specimens. For some of the species, sub-species have been identified in the literature (see, i.e., Pugh 1993); however, since the validity of the sub-species is unclear, their differentiation was not carried out in the present study. The juvenile individuals (nymphs), which constituted a vast majority of all registered specimens of the Oribatida, most likely belong to *Alaskozetes antarcticus*.

All oribatid species identified in the present project as well as their position within the systematic tree are given in the following list. Systematics follow Norton & Beham-Pelletier (2009).

Ameronothroidea
Ameronothridae
Alaskozetes antarcticus (Michael, 1903)
Halozetes belgicae (Michael, 1903)

Oppioidea
Oppiidae
Globoppia loxolineata (Wallwork 1965)

Brachychthonioidea
Brachychthoniidae
Brachychochthonius sp.
Liochthonius cf. *mollis* (Hammer, 1958)

Table 9: The literature used for the identification of the species of Oribatida recorded in the present Antarctic material.

Species	Determination according to
Alaskozetes antarcticus	Wallwork 1962
Globoppia loxolineata	Wallwork 1965
Halozetes belgicae	Wallwork 1965
Liochthonius cf. *mollis*	Hammer 1958

3.4.2.2.2.1 Indigenous antarctic species

Alaskozetes antarcticus (Michael, 1903)

A. antarcticus is most likely of maritime Antarctic origin, where it is widespread (Convey et al. 2000a, Fig. 139). The species is distributed with many sub-species circumpolarly (Dalenius 1965), where it occurs not only in the maritime Antarctic, but also in continental Antarctica, the subantarctic as well as the southern tip of New Zealand and possibly also Australia (Davies et al. 1997, Marshall et al. 1999, Pugh 2003, Starý & Block 1998, Australian Data Research Center). In localities where it occurs, *A. antarcticus* is often broadly distributed; for instance, it was found on all of the South Sandwich Islands, occurring in more than half of the samples taken on these islands (Convey et al. 2000a).

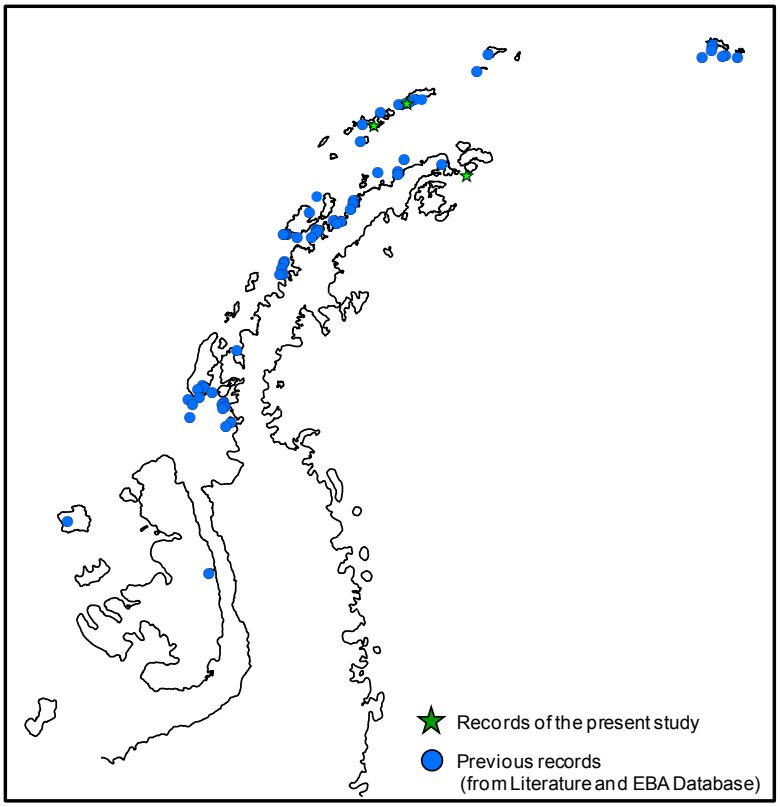


Fig. 139: Records of *A. antarcticus* in the maritime Antarctic. Sources: acarological and soil faunal literature from the Antarctic (as far as available to the authors) as well as the Biodiversity Database of the Australian Antarctic Data Centre.

The species is commonly found at low attitudes and in the littoral zone of coastal areas (Goddard 1979b, Richard et al. 1994, Convey & Quintana 1997, Convey & Smith 1997, Pugh 2003), at times also in waterlogged sites where it can survive immersion for up to nine months (Richard et al. 1994, Convey & Quintana 1997). It has been found in a wide range of habitats: under and on stones, in algal mats (mostly *Prasiola crispa*), on lichens, mosses as well grass (Gressitt et al. 1963, Dalenius 1965, Gressitt 1967, Goddard 1979b, Richard et al. 1994, Tilbrook 1967b, Convey & Quintana 1997). It is usually found in sites enriched in organic material, i.e., bird nests, guano, Penguin rookeries etc. (Gressitt 1967, Goddard 1979b, 1979c, Convey & Quintana 1997, Davies et al. 1997).

The species often occurs in strong aggregations (Gressitt et al. 1963, Strong 1967, Tilbrook 1967, Goddard 1979b,Convey & Smith 1997). The species often overwinters in these dense aggregations, whereby all still life stages can overwinter, females often overwinter with eggs, and the overwintering sites can be used for many years (Strong 1967). Although *A. antarcticus* can occur in moist sites, it is often been described as preferring drier habitats (Gressitt et al. 1963, Tilbrook 1967b). The species is been recorded as occurring in lesser densities in warmed soils (i.e. fumuroles) (Convey et al. 2000a). *A. antarcticus* is a detritus feeder and scavenger, feeding on organic detritus, lichens and algae (Strong 1967, Goddard 1979c).

A. antarcticus on its own did not correlate consistently with any of the background habitat parameters (Appendix 4, Table A4-3). However, when the species was analyzed together with the nymphs (most of which in all likelihood belong to this species), it showed in both years a positive relationship to amount and quality of the organic material of the soil substrates. As

with total densities, the species correlated 2010 positively to soil moisture and in 2011 positively to vegetational cover and negatively to soil temperature and soil pH-value.

In 2010 the species was found significantly more abundant in the anthropogenically non-influenced areas (Fig. 140; Appendix 5, Table A5-6). In 2011, however, the human impact on the densities of *A. antarcticus* depended on locality, being sometimes more abundant in influenced areas and other locations in the non-influenced areas (Fig. 140). Due to these locality specific tendencies, the human influence on its densities were statistically not significant in 2011 or when both years were analyzed together.

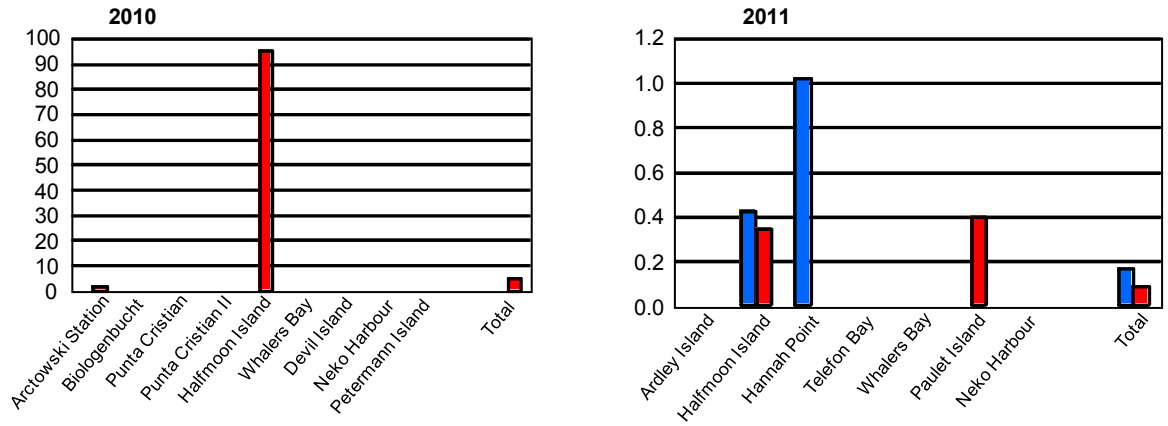


Fig. 140: Total densities of *A. antarcticus* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analyses could specify the results of the variance analyses of the species more precisely. In 2010 the species was more abundant in non-influenced areas (Fig. 141; Appendix 6, Table A6-3). It was significantly more abundant in areas of medium vegetational cover. The human influence on its abundances were highest in samples of medium vegetational cover, as shown by the statistical interaction between human influence and vegetational cover. In 2011, on the other hand, *A. antarcticus* was actually significantly more abundant in the human influenced areas (Fig. 142; Appendix 6, Table A6-3). In this year the species was again significantly more abundant the higher the degree of vegetational cover was. Again, the human influenced the strongest in samples of medium vegetational cover (significant interaction between human influence and vegetational cover).

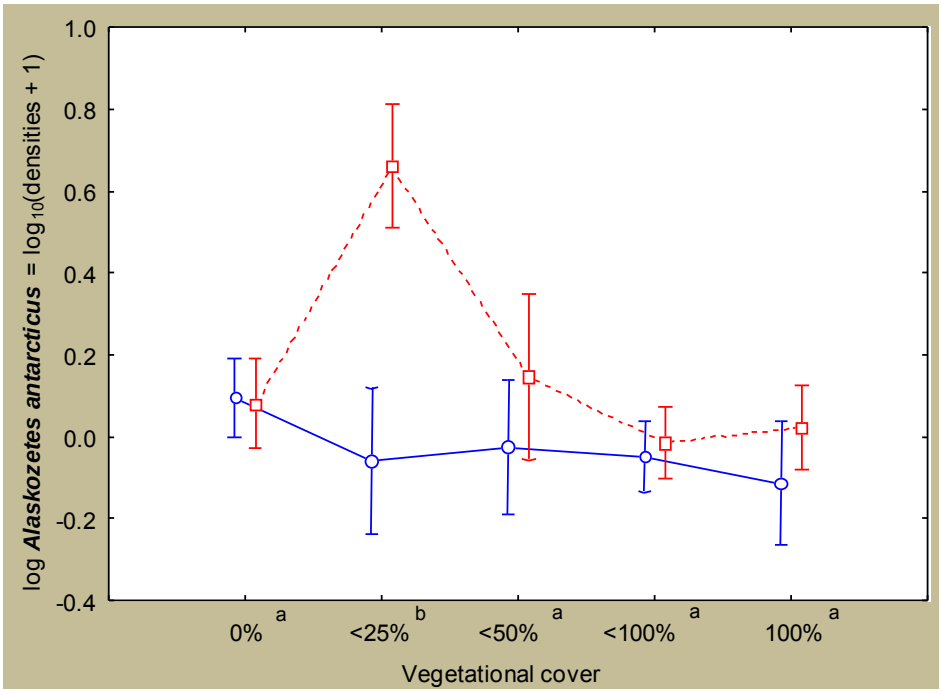


Fig. 141: Results of the covariance analysis (ANCOVA) of the densities of *A. antarcticus* recorded in 2010 after filtering out various background habitat parameters. Densities in anthropogenically influenced areas in blue and in non-influenced areas in red. Figure explanations as in Fig. 132.

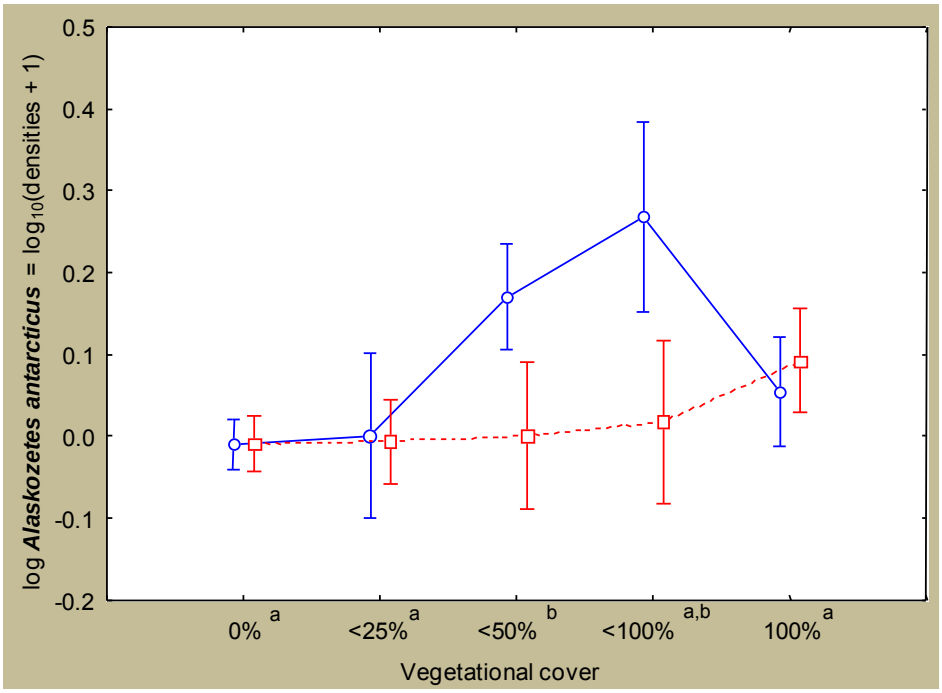


Fig. 142: Results of the covariance analysis (ANCOVA) of the densities of *A. antarcticus* recorded in 2011. Densities in anthropogenically influenced areas in blue and in non-influenced areas in red. Figure explanations as in Fig. 132.

Globoppia loxolineata (Wallwork 1965)

G. loxolineata has been hitherto recorded mostly from the Antarctic Peninsula and the neighboring islands (Starý & Block 1998; Fig. 143); where according to the frequency of its records it appears to be not seldom. Further records are also from the Antarctic South Shetland and South Orkney Islands, the continental Antarctic as well as the Subantarctic, i.e., Heard

Island in the Indian Ocean (Pugh et al. 1994, Block & Starý 1996, Starý et al. 1997, Convey et al. 2000a, Australian Data Research Center).

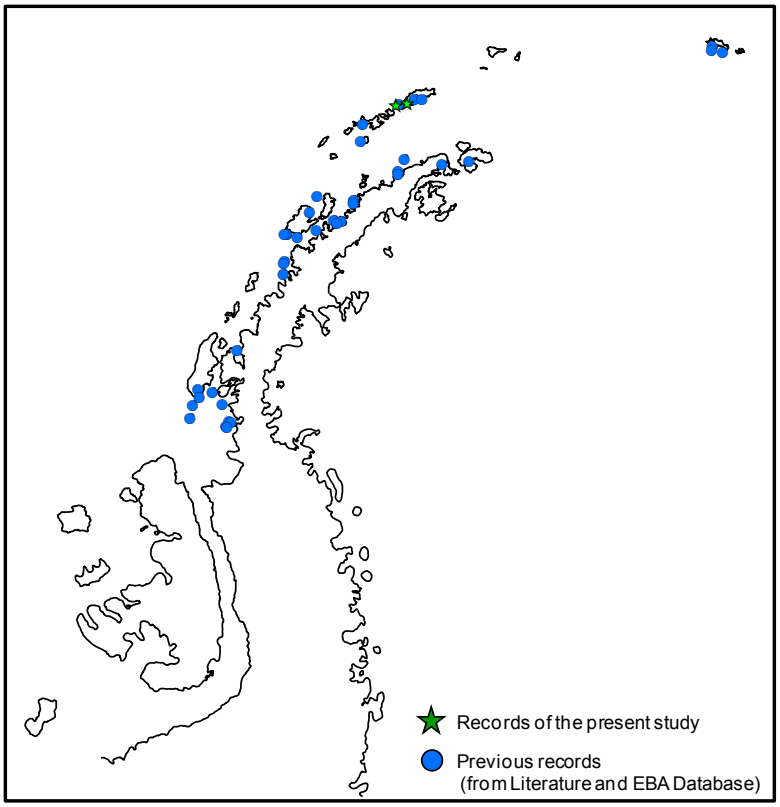


Fig. 143: Records of *G. loxolineata* in the maritime Antarctic. Sources as in Fig. 139.

The species has been found in a wide range of habitats, from under and on stones, in algal mats, lichens, mats of mosses and grass, to nests of birds etc. (Gressitt 1967, Goddard 1979b, Convey & Quintana 1997, Pugh 2003). It apparently occurs less than the previous species in wet sediments, but seems also avoid dry barren areas (Goddard 1979b, Convey & Quintana 1997). It is also usually been found more as scattered individuals and seems not to form large aggregations as does the previous species (Strong 1967, Goddard 1979b). The maximum densities of the species can be found in early spring and late summer (Goddard 1979b).

The main overwintering stage it appears to be middle juvenile stages (deutonymphs), but otherwise seems to have no clue the yearly cycle; it does not overwinter in aggregations as does the previous species (Strong 1967, Goddard 1979b). *G. loxolineata* is most likely scavenger, also feeding on dead plant material as well as also fungal hyphae (Gressitt 1967, Strong 1967).

This species was only recorded from two locations on the Fildes Peninsula of King George Island and only in 2010 (cf. Appendix 3, Table A3-3). Due to it only being registered in very few locations, no significant correlations between its densities and background habitat factors could be discerned.

In one of the locations (Punta Christian) it was found in higher abundances in the non-influenced areas (Fig. 144); however, due to high sample-to-sample variability (many samples without specimens) these results were statistically not significant. No results were obtained from the covariance analyses.

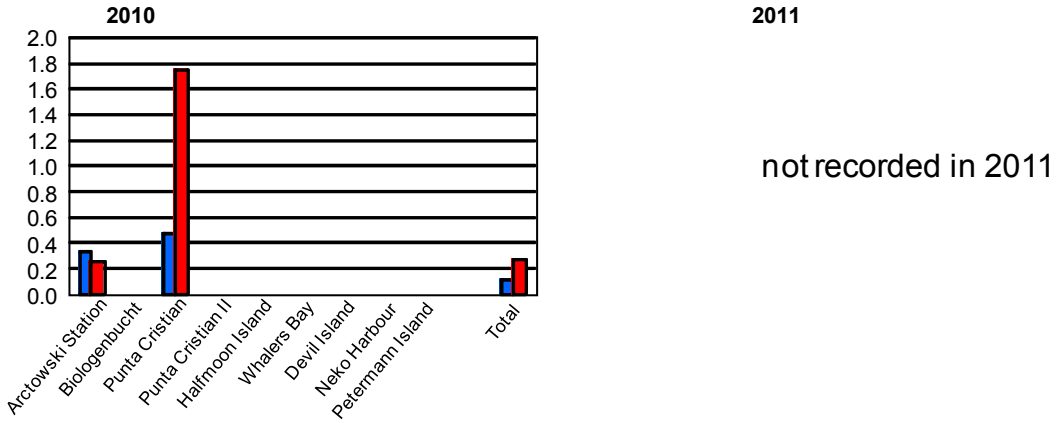


Fig. 144: Total densities of *G. loxolineata* (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

Halozetes belgicae (Michael, 1903)

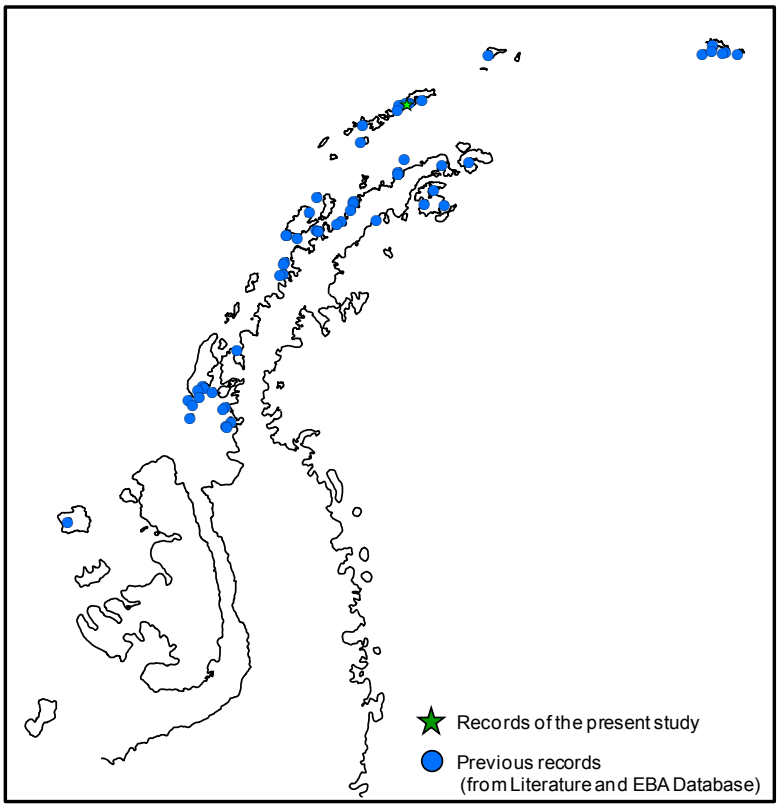


Fig. 145: Records of *H. belgicae* in the maritime Antarctic. Sources as in Fig. 139.

This species is with many subspecies distributed circumpolarly widely in the Antarctic and Subantarctic (Pugh 2003, Pugh et al. 1994, Starý & Block 1998, Convey et al. 2000a, Sanyal 2004). It is also one of the most widespread oribatid mite species in the maritime Antarctic (Fig. 145). Despite its being so widespread, little has been recorded about the habitat preferences and feeding habits of *H. belgicae*. It is been found in the littoral and supralittoral zones, on algae, lichens, as well as in mosses (Gressitt 1967, Tilbrook 1967b, Pugh et al. 1994, Pugh 2003).

As *H. belgicae* was only recorded in few specimens in only one location (Appendix 3, Table A3-3), no statistical analysis of its relationship with habitat background parameters or of human influence on its densities could be undertaken.

Liochthonius cf. *mollis* (Hammer, 1958)

L. mollis is hitherto known from the Subantarctic islands in the Indian and Atlantic oceans, southern South America as well as the Antarctic South Shetland Islands (Starý & Block 1998). The species' distribution apparently just barely reaches the margin of the Antarctic zone; the study sites in the present project most likely represent the southern border of its distributional area. It has thusly only rarely been recorded in the Antarctic (Fig. 146), where it has been found among vegetation (Pugh 2003). Little else is known about its habitat preferences, ecological tolerances or biology.

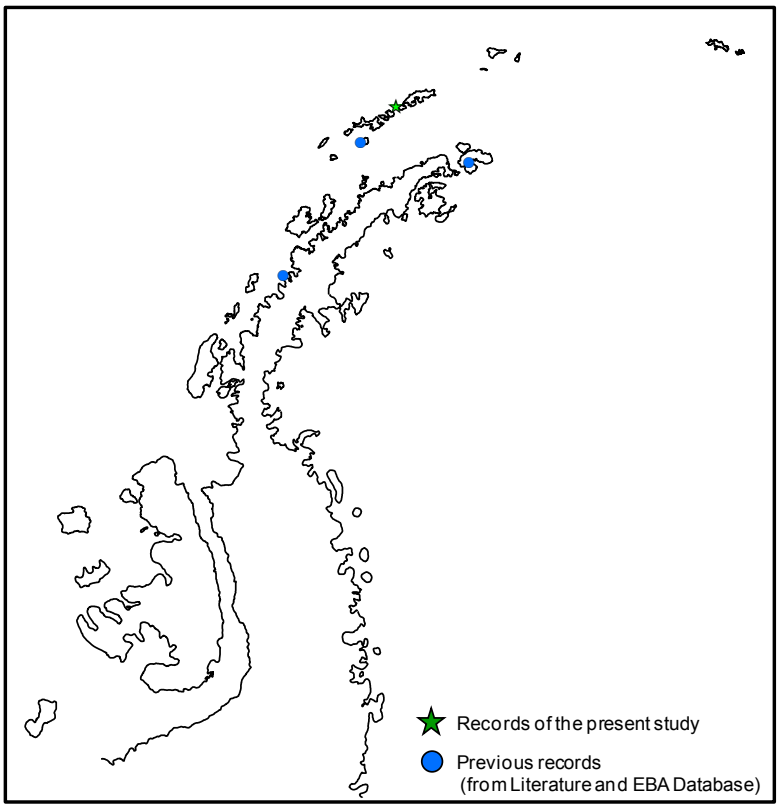


Fig. 146: Records of *L. mollis* in the maritime Antarctic. Sources as in Fig. 139.

In the present project, it was only recorded once as single individuals from Punta Christian (Fildes Peninsula, King George Island). Therefore, no statistical analysis could be undertaken for the species.

3.4.2.2.2.2 Potentially introduced (non-native) species

No non-indigenous species of the Oribatida were recognized.

3.4.2.3 Gamasina

3.4.2.3.1 General community parameters

A total of 131 individuals of the Gamasina were recorded in the present study, 88 in the year 2010 and 43 in 2011. In many localities no gamasine mites at all were registered. In those localities where Gamasina were found, maximum densities reached over 70 individuals per cm^3 substrate in 2010 and 30 individuals per cm^3 substrate in the year 2011. As in the other microarthropod groups, significant differences in the total densities of Gamasina could also be discerned between the various locations (Fig. 147; Appendix 5, Table A5-1). In the year 2010, the significant differences were mostly between localities with Gamasina and those without these mites, while in 2011 significant differences also existed between localities in which gamasine mites were found. Although a few more individuals were found in the year 2011 than 2010, no significant differences in the densities found in those localities sampled in both study years could be discerned.

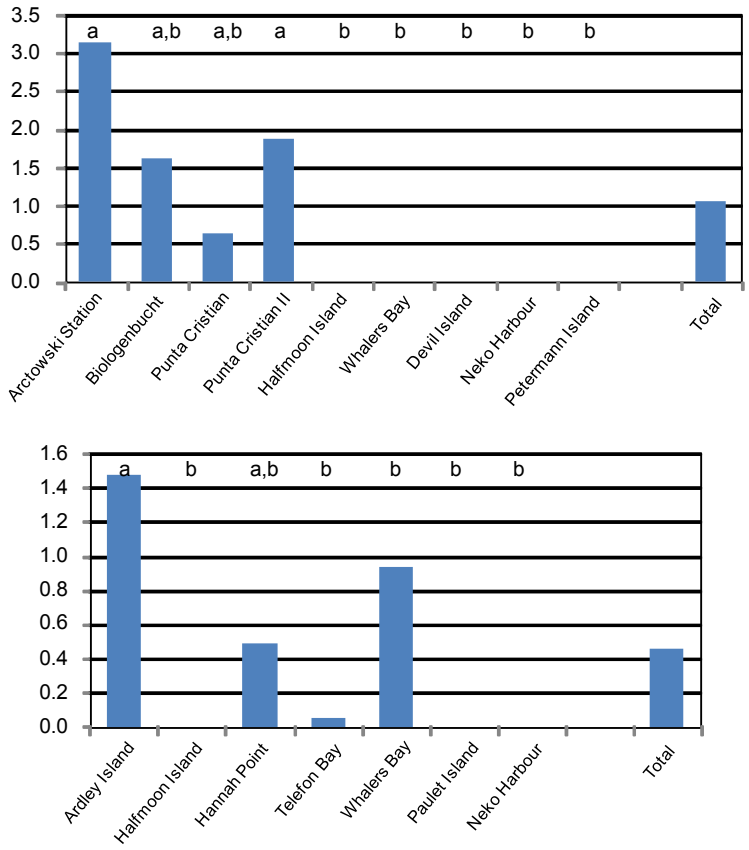


Fig. 147: Average total gamasine densities (in individuals per 100 cm^3 substrate) recorded in the various studied localities in 2010 (above) and 2011 (below). Different letters denote significant differences (= densities in localities with the same letter are statistically *not* different from one another). Note the different scales of the y-axis for the two years.

Significant correlations between gamasine total densities and parameters representing locality as well as a positive relationship between the densities and vegetational cover could be discerned in both years (Appendix 4, Table A4-4). In 2010, the densities of Gamasina correlated

positively to amounts and quality of organic material, while in 2011 the densities correlated positively to soil moisture.

Concerning an anthropogenic influence on total gamasine densities, in most localities higher densities were found in the non-influenced areas (Fig. 148). However, mostly due to many samples containing no Gamasina at all, these results were statistically not significant in the year 2010, only showed a statistical tendency in the year 2011, but were highly significant when both years were analyzed together (Appendix 5, Table A5-6).

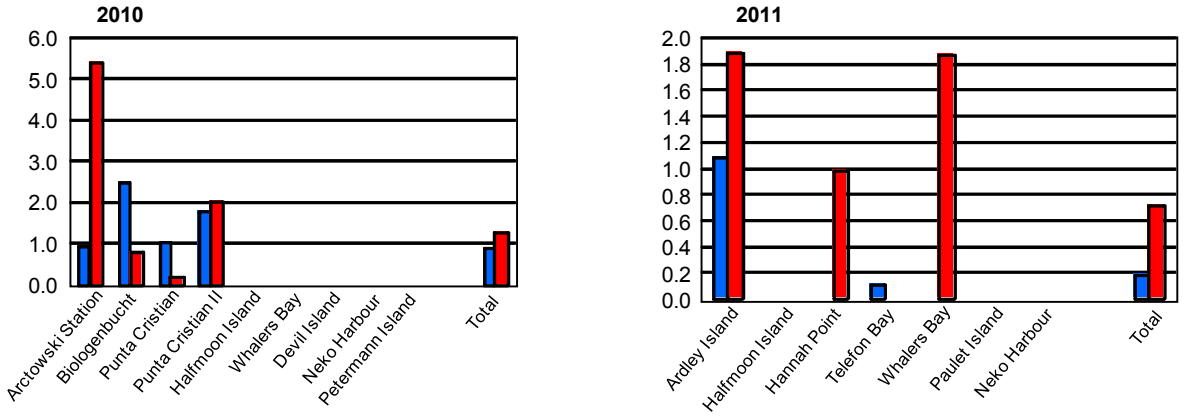


Fig. 148: Total gamasine densities (in individuals per 100 cm^3 substrate) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the study year 2010 (left) and 2011 (right).

Filtering out the background habitat factors, the covariance analyses of the gamasine densities could primarily show that in the year 2010 total densities became larger with increasing vegetational cover (Fig. 149; Appendix 6, Table A6-4). In year 2011, according to this analysis, densities were also significantly larger in anthropogenically non-influenced areas (Fig. 150).

Species richness of the Gamasina was the lowest for all microarthropod groups. Four species were registered in total, two in the year 2010 and three in 2011. Although total species richness was low, a significant difference in the average number of species per sample could be discerned between the various locations in both years (Fig. 151; Appendix 5, Table A5-1). No year-to-year difference in gamasine species richness could be discerned.

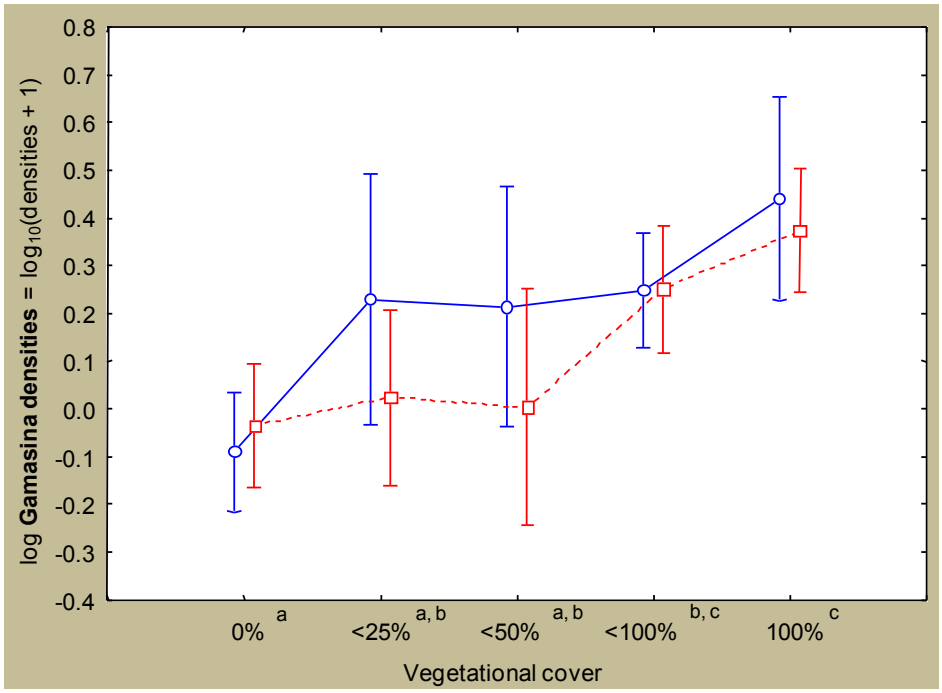


Fig. 149: Results of the covariance analysis (ANCOVA) of total densities of Gamasina recorded in 2010. Densities in anthropogenically influenced areas in blue and in non-influenced areas in red. Different letters denote significant gamasine density differences between vegetational cover categories. Negative values (y-axis) are statistical artefacts and result from the logarithm of very low densities.

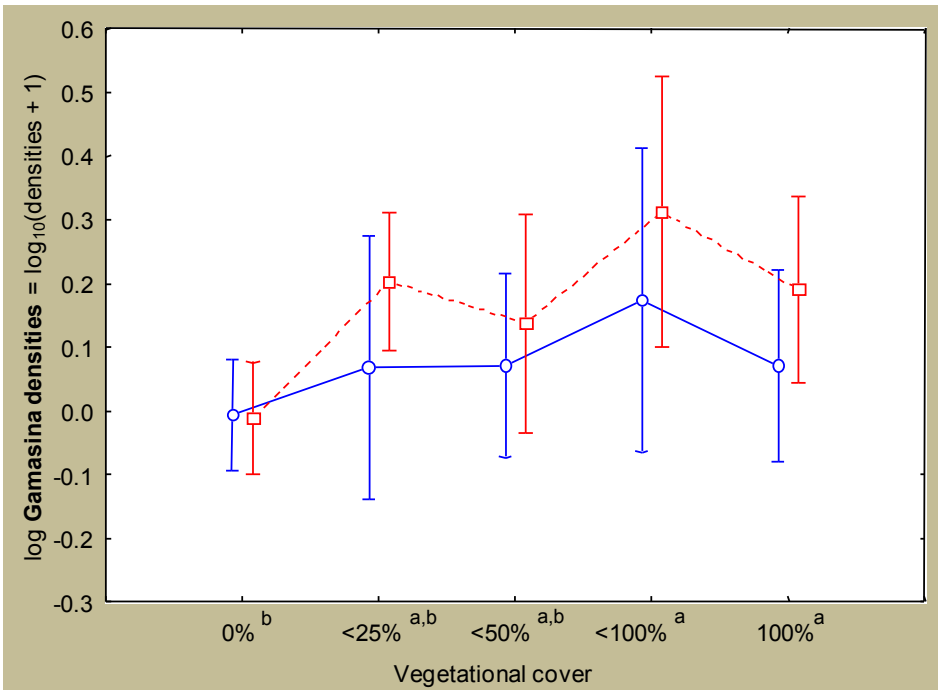


Fig. 150: Results of the covariance analysis (ANCOVA) of total gamasine densities recorded in 2011. Figure explanation as in Fig. 149.

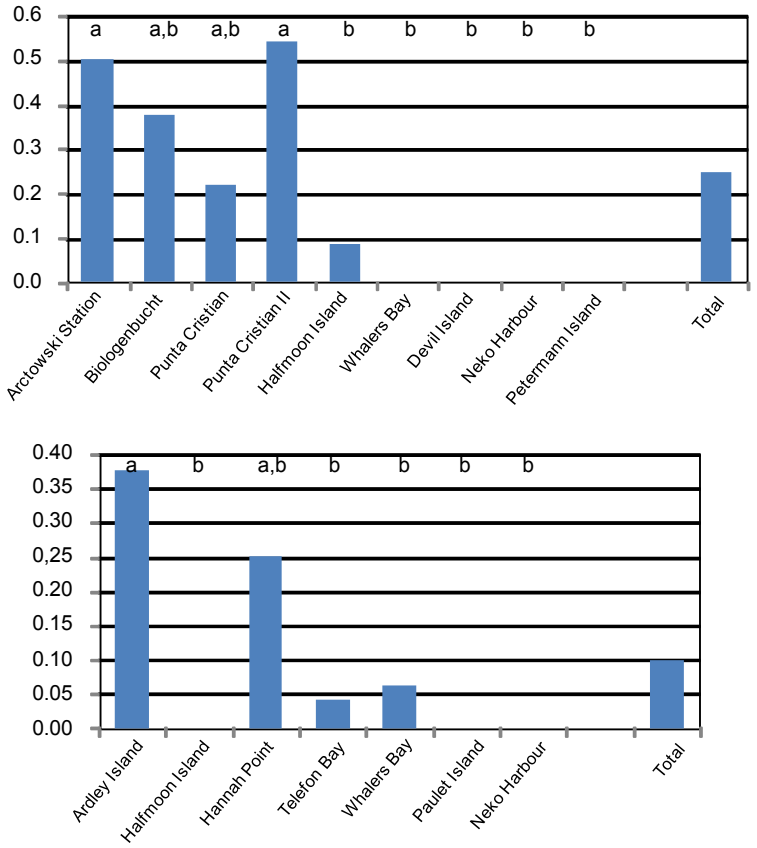


Fig. 151: Average gamasine species richness (average number of species per sample) recorded in the various studied localities in 2010 (above) and 2011 (below). Figure explanation as in Fig. 147.

Species richness of the Gamasina correlated significantly with parameters representing locality as well as (positively) to vegetational cover and soil moisture in both years (Appendix 4, Table A4-4). Only in the year 2010 did species richness correlate positively with amounts and quality of organic material in the soil substrates.

Whether more species were recorded in anthropogenically influenced or not influenced areas depended on locality, where the different localities showed conflicting results (Fig. 152). Thus, overall no significant human influence on species richness could be discerned, although a slight statistical tendency towards higher species richness in non-influenced areas could be distinguished (Appendix 5, Table A5-6).

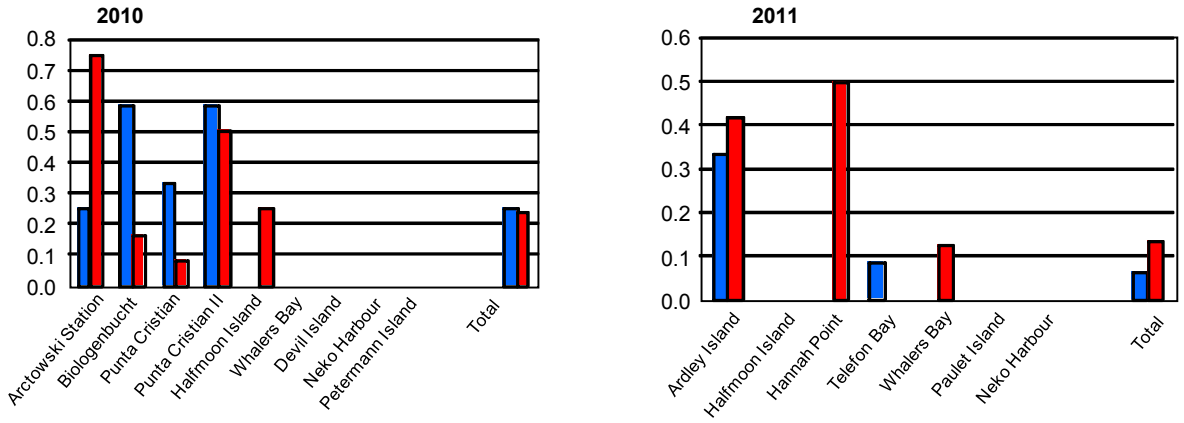


Fig. 152: Average species richness (average number of species per sample) of the Gamasina recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

The covariance analysis primarily showed significantly higher species richness with increasing vegetational cover in the study years 2010 and 2011 (Figs 153 and 154; Appendix 6, Table A6-4). Furthermore, according to this analysis species richness was significantly higher in non-influenced areas in the year 2011 (Fig. 154).

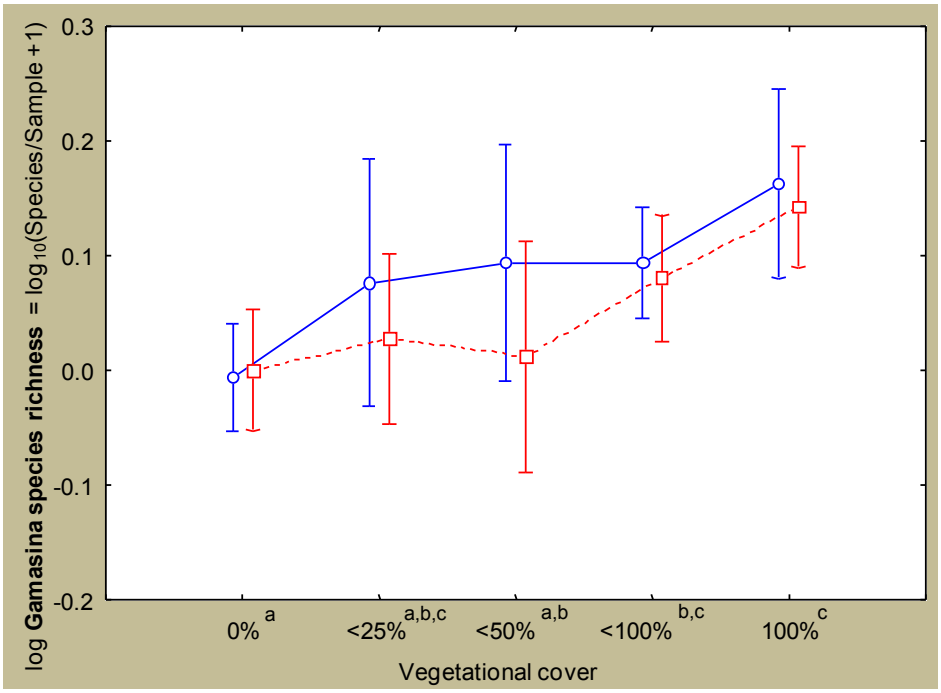


Fig. 153: Results of the covariance analysis (ANCOVA) of species richness of the Gamasina recorded in 2010. Figure explanation as in Fig. 149.

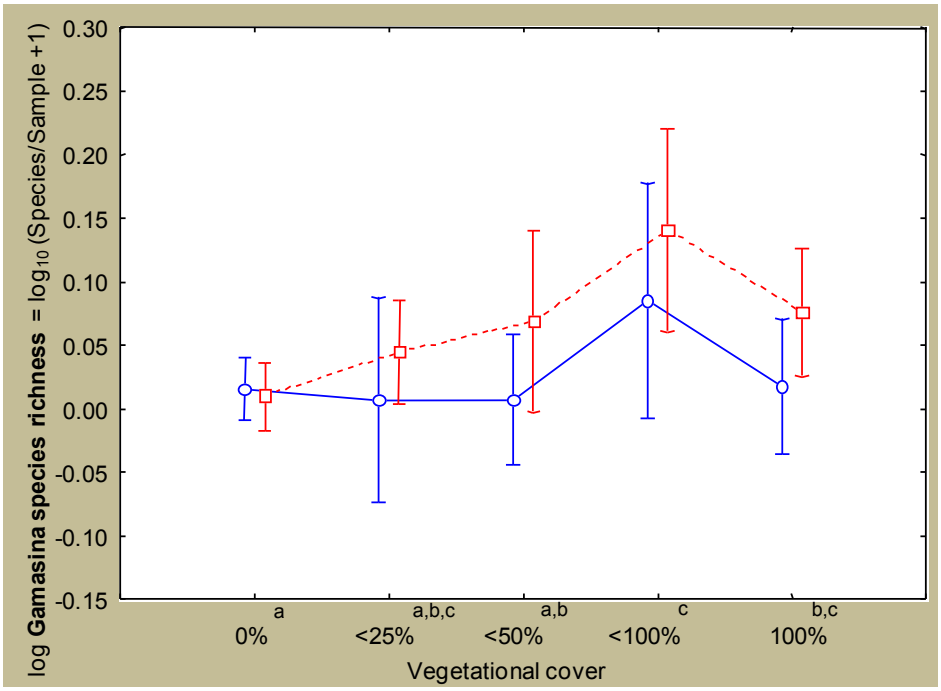


Fig. 154: Results of the covariance analysis (ANCOVA) of average gamasine species richness recorded in 2011. Figure explanation as in Fig. 149.

3.4.2.3.2 Results and descriptions of the determined species

Altogether four species of the Gamasina were determined in the present study. Unfortunately, due to few specimens being available for deeper taxonomical study, two of the species could not be determined to species level because in one species only two larvae (Gen. sp. 1) were recorded, and in the other species only one protonymph (Gen. sp. 2). For a secure determination, in most cases adults are necessary. The determined species and their systematical position are given in the following list. The average total densities of the species in the various localities in Appendix 3, Table A3-3. Systematics follow Krantz & Walter (2009) in the higher taxonomic groups and Karg & Schorlemmer (2009) at the family and generic level.

Ologamasidae
Hydrogamasellus racovitzai (Trouessart, 1903)
Hydrogamasellus sp. 2 (only two larvae)

Parasitidae
Parasitus tarsispinosus Hunter, 1967

(unknown)
Gen. sp. 2 (only one protonymph)

3.4.2.3.2.1 Indigenous Antarctic species

Hydrogamasellus racovitzai (Trouessart, 1903)
Syn. *Gamasus racovitzai* Trouessart, 1903
Syn. *Gamasellus racovitzai* (Trouessart, 1903)
Syn. *Cyrtolaelaps racovitzai* (Trouessart, 1903)

H. racovitzai is a large, conspicuous mite, which has been found to be widely distributed throughout the maritime Antarctic (Fig. 155). However, the species has also often been recorded in the Subantarctic (Pugh et al. 1994, Convey & Quintana 1997). This species has been recorded from a wide range of habitats, e.g., on or under stones, in algal mats, on lichens,

mosses, grass as well as in bird nests (Dalenius 1965, Tilbrook 1967b, Richard et al. 1994, Convey & Quintana 1997, Convey & Smith 1997). In mosses it apparently occurs in the upper layers (0-3 cm) (Goddard 1979a). In which habitat type the maximum densities are found seems to be more dependent on location than habitat type; the most common factor apparently being availability of abundant prey (Strong 1967).

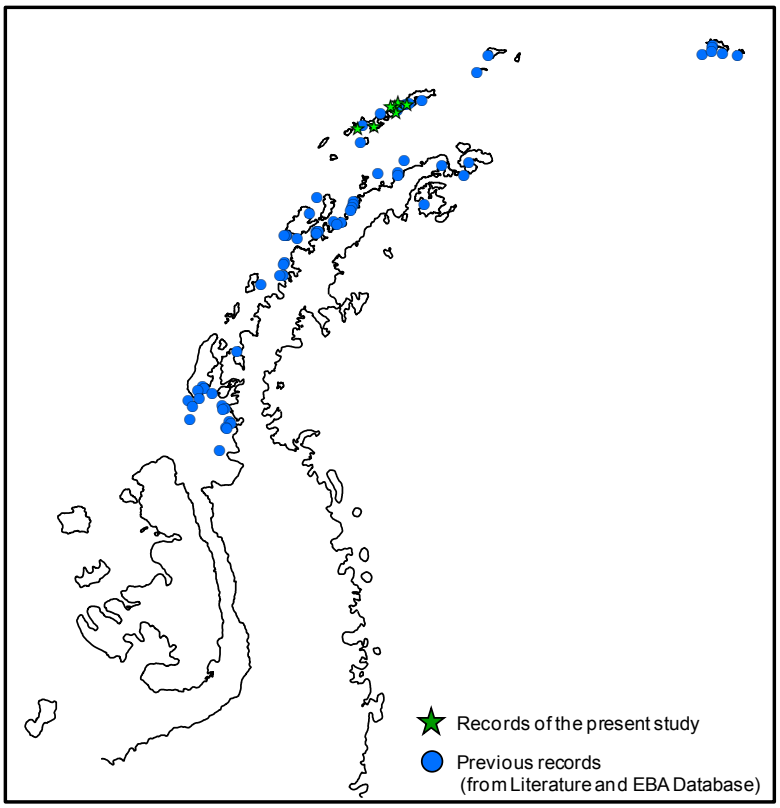


Fig. 155: Records of *H. racovitzai* in the maritime Antarctic. Sources: acarological and soil faunal literature from the Antarctic (as far as available to the authors) as well as the Biodiversity Database of the Australian Antarctic Data Centre.

H. racovitzai is a very active predator, feeding mostly on Collembola and mites (Gressitt 1967, Strong 1967, Goddard 1979c). It shows a clear circadian activity pattern, being more active around midnight; it has been shown that the species has a weak negative relationship to temperature (Burn & Lister 1988). These factors may assist its ability to catch Collembola, which may be slowed by lower temperatures.

This species was the most abundant taxon of the Gamasina found in the present study, where it was recorded exclusively from the South Shetland Islands (Fig. 155). Its correlations with habitat parameters generally followed those of total gamasine densities (Appendix 4, Table A4-4). In both years this species correlated consistently with latitude (as an indicator of locality) as well as positively with vegetational cover. In 2010 this species' densities correlated positively with quantities of organic matter in the sampled substrates, and in 2011 positively with soil moisture.

Although *H. racovitzai* was generally found in higher individual numbers in the anthropogenically non-influenced areas (Fig. 156), no significant human impact beyond a slight tendency in 2011 could be statistically discerned. No results were obtained with the covariance analyses of this species' densities.

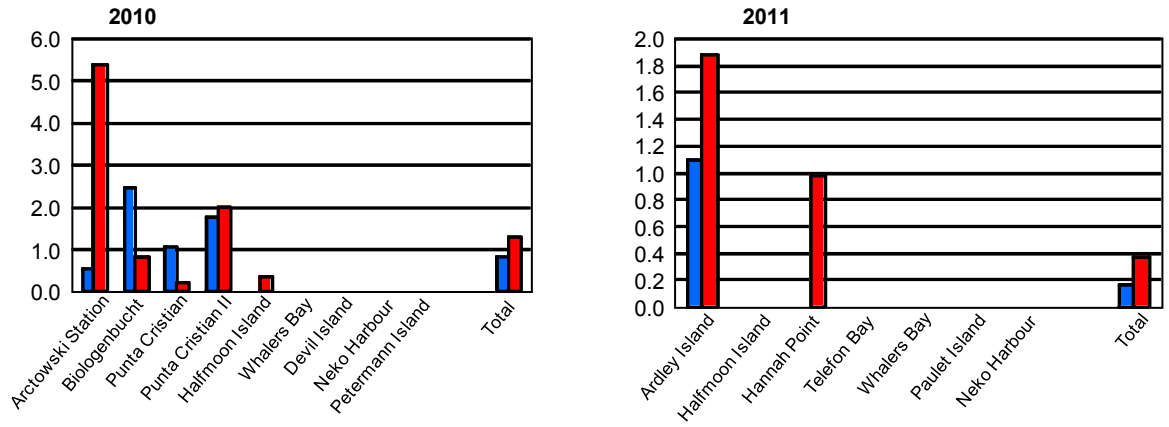


Fig. 156: Total densities of *H. racovitzai* (in individuals 100 cm^{-3}) recorded in the anthropogenically influenced (blue) and non-influenced (red) areas in the two study years.

In the present study, a further taxon within this genus was recorded, *Hydrogamasellus* sp. 2. Its identity at the species level is still unknown and it possibly represents a new species. A clarification of this requires further taxonomical study. Due to the very sporadic occurrence of this taxon in the samples with only very few individuals in total, no assessment of its distribution, its habitat preferences, possible anthropogenic impacts on its densities or its status (native or non-native) can be provided. Representatives of this taxon were recorded near Arctowski Station at Admirality Bay in King George Island in the year 2010. Similar can be said regarding a second, also not specifically identifiable taxon: Genus sp. III was found in a single individual in Telefon Bay on Deception Island.

Parasitus tarsispinosus Hunter, 1967

The species P. tarsispinosus has only been recorded once the maritime Antarctic, where it was found "under wood on an ash plain" on Deception Island (Hunter 1964, Downie 2002 in Valencia & Downie 2002). In the present project, it was also only found once in multiple individuals of two samples on Deception Island (Fig. 157). Its record in the present study confirms its occurrence in viable populations on Deception Island.

This species occurred in far too few samples or localities to be able to carry out a statistical analysis of its relationship to habitat parameters, of its distribution within the study localities or of an influence of human activities on its densities.

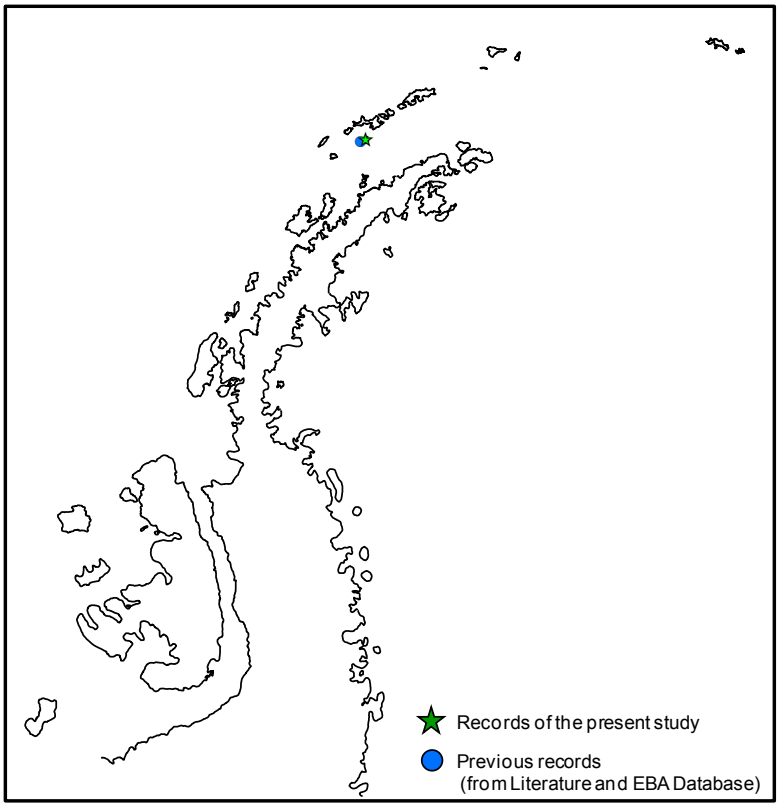


Fig. 157: Records of *P. tarsispinosus* in the maritime Antarctic. Sources as in Fig. 155.

3.4.2.3.2.2 Potentially introduced (non-native) species

No non-indigenous species of the Gamasina were recognized.

3.5 Microfauna

3.5.1 Nematoda (Roundworms)

The phylum Nematoda (roundworms) comprises microscopically small, unsegmented and usually radially symmetric worms, which have very successfully colonized all at least sporadically moist habitats of our planet. Nematodes live as parasites in plants and animals and occur free-living in the pore space of soils and all limnic and marine sediments as well as in mosses, lichens and algal mats. Only air and free water are not actively inhabited by nematodes, although these habitats are passively "used" for distribution (passive wind and water dispersal).

The first Antarctic nematodes were collected during the Belgica expedition (1897-1899) by the Romanian biologist Emil Racovita in small meltwater accumulations near Beneden Head. These individuals were later described by de Man (1904) as *Plectus antarcticus, Plectus belgicae* and *Mononchus* sp. (later placed in *Coomansus gerlachei*). Today 54 nematode species are known from Antarctica and further new species are constantly being discovered. Approximately 85% of the species reported from the Antarctic are endemic, i.e. have only been found in Antarctica (Andrássy 2008). From these, 32 species have been recorded from the climatically milder Maritime Antarctic (including the Antarctic Peninsula) and only 22 species from the much larger, but climatically harsher continental Antarctic (Andrássy 2008). Distributional overlap -

species that occur both in Maritime and Continental Antarctica - practically does not exist (but see Maslen & Convey, 2006). Depending on species, soil-living nematodes feed on bacteria, fungi, algae, dead organic material, plant-root sap, protozoa or other soil animals (Yeates et al. 1993). Due to the usually lacking vegetation and the low soil nutrient content, nematode bacterial and algal feeders play a major role in Antarctic soil food webs (Andrássy 1998).

As aquatic life forms, nematodes are dependent upon the availability of water for their various activities, e.g., mobility, feeding, growth, respiration and reproduction. Since terrestrial habitats periodically or regularly dry out, many soil-born nematode species are capable of anhydrobiosis, an ametabolic state of dormancy, in which an individual can survive adverse conditions for many years (Wharton, 2002). When sufficient water returns, the individual becomes active within minutes or hours. Although large amounts of water exist in Antarctica, this water is mostly biologically not available, since it is bound in ice, snow, clay or soil-bound organic material. The availability of water therefore plays a major role in the distribution of terrestrial nematode species in Antarctica (Wharton, 2003). In the Dry Valleys, 30 to 80% of all nematodes are present in a state of anhydrobiosis (Treonis et al. 2000). Desiccation-induced anhydrobiosis also protects these species from other environmental stressors, e.g., extremely low temperatures. However, even without a foregoing anhydrobiosis, many Antarctic nematode species are capable of surviving temperatures as low as -30°C and intracellular ice crystals without damage (Smith et al. 2008, Wharton et al. 2005). In light of these survival strategies, it is not surprising that nematodes represent the individual- and species-richest animal group in the terrestrial Antarctic.

3.5.1.1 General faunistical parameters

From a total of 328 soil samples in the present study, more than 255,000 nematode individuals were extracted and quantified. From these, a total of 18,322 (maximally 100 individuals per sample) were determined taxonomically. Average densities of between 0.8 (Neko Harbour) and 11,344 (Arctowski Station) individuals per 100 cm^3 substrate were detected (Fig. 158). Species richness per sample ranged between 0.2 (Neko Harbour) and 11.3 (Punta Christian I). In both study years, the individual and species richness of the nematodes differed significantly among the various localities (Appendix 5, Table A5-1), whereby in 2010 the northernmost study sites on King George island showed the highest densities (Arctowski Station) and species richnesses (Biologenbucht and Punta Christian) (Fig. 158). In contrast, in 2011 the highest densities were found in the soils from Paulet Island, one of the more southern islands in the present study (Fig. 158). However, these high total densities were caused by an extremely high population growth of a single species, *Rhomborhabditis* cf. *teres*. In this year, the largest species richness (7.5 species per sample) was found on Ardley Island (Fig. 158).

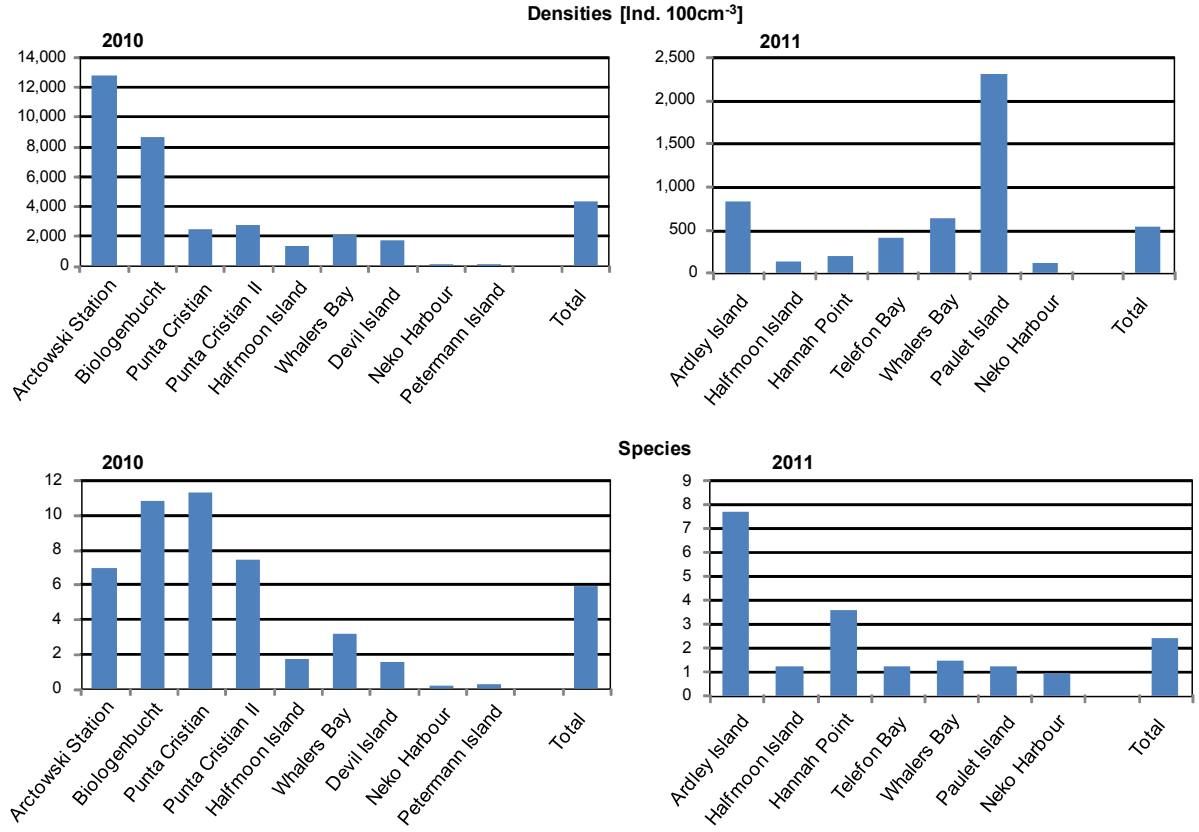


Fig. 158: Nematode densities (individuals per 100 cm^3 substrate) and average species number per sample recorded in the various study sites in 2010 and 2011. Study sites are ordered from left to right by increasing southern latitude. Different letters denote significant differences between sites, i.e., the densities or species richness in sites with a common letter were statistically not different. Please note the different scales of the y-axes.

The anthropogenically influenced areas showed significantly lower total densities than the non-influenced areas, both when the data of the individual study years were analyzed separately and when they were analyzed together (Fig. 159; Appendix 5, Table A5-7). On the other hand, human activity apparently had no influence on the determined number of nematode species per sample.

Vegetational cover had a significant positive influence on total individual densities, especially in 2010 (Fig. 160; Appendix 6, Table A6-5). This effect was also present in 2011, but quantitatively less so. Nematodes in soil samples with a vegetational cover > 0% were individual richer than in samples without vegetation. When considering the nematode numbers among the different levels of vegetational cover, human trampling alone had no significant effect. However, in 2010, a significant statistical interaction between human influence and vegetational cover was detected (Fig. 160; Appendix 6, Table A6-5), which indicates that a stronger anthropogenic influence existed when vegetational cover was higher.

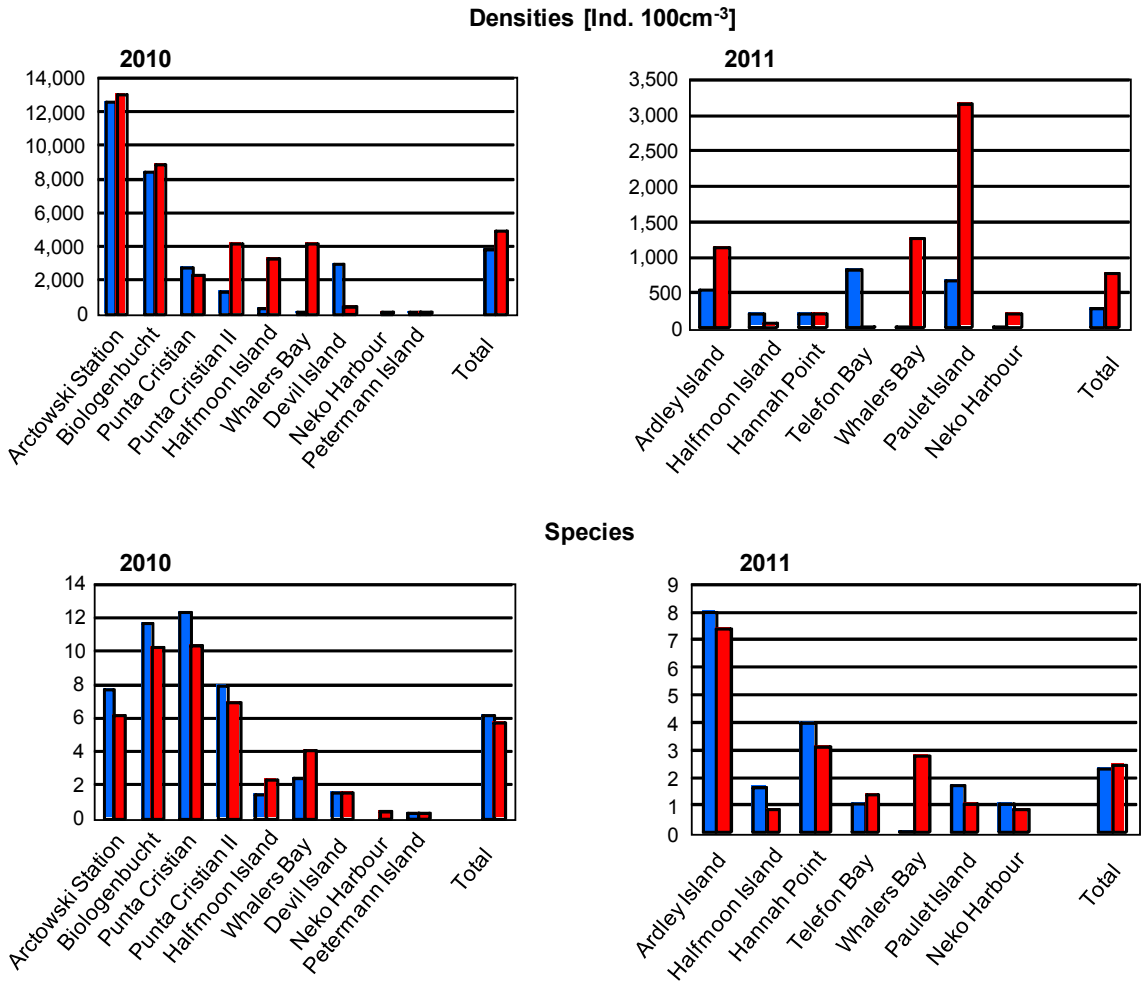


Fig. 159: Nematode densities (individuals per 100 cm^3 substrate) and average number of species per sample in the various study sites, separately for anthropogenically influenced (blue) and non-influenced (red) areas, as well as for 2010 and 2011. Please note the different scales of the y-axes.

The degree of vegetational cover also had a significant influence on the number of species per sample (Fig. 161; Appendix 6, Table A6-5). Soil samples without vegetation were species poorer than samples with vegetation. Human trampling alone showed no significant effect. However, an anthropogenic effect was dependent upon the degree of vegetational cover, as indicated by the significant statistical interaction between anthropogenic influence and vegetational cover (Fig. 161).

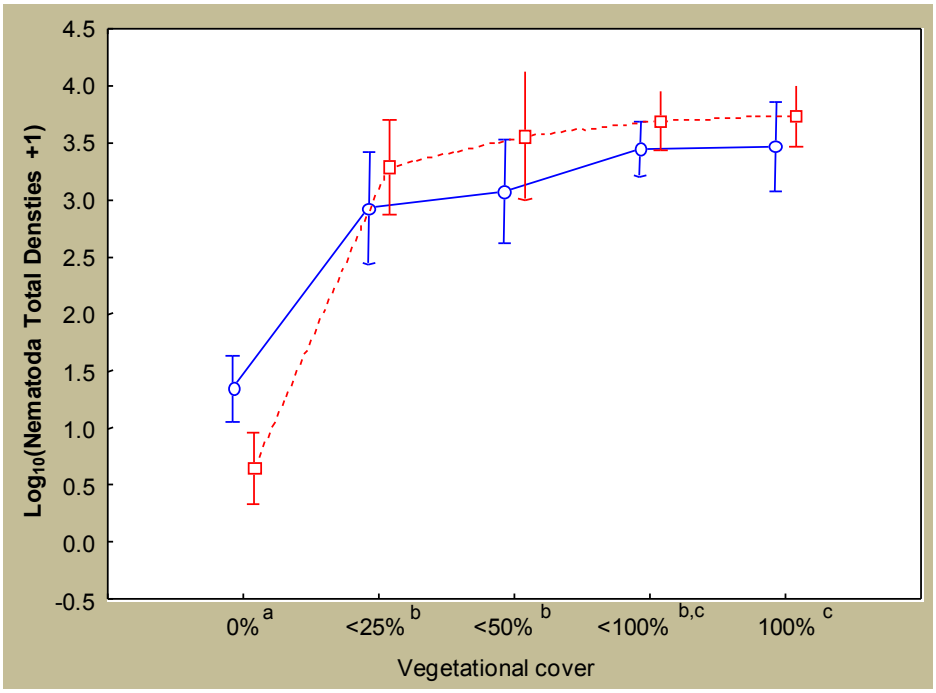


Fig. 160: Total nematode individual densities in 2010 in anthropogenically influenced (blue) and non-influenced (red) areas in relation to vegetational cover. Different letters denote significant density differences between degrees of vegetational cover.

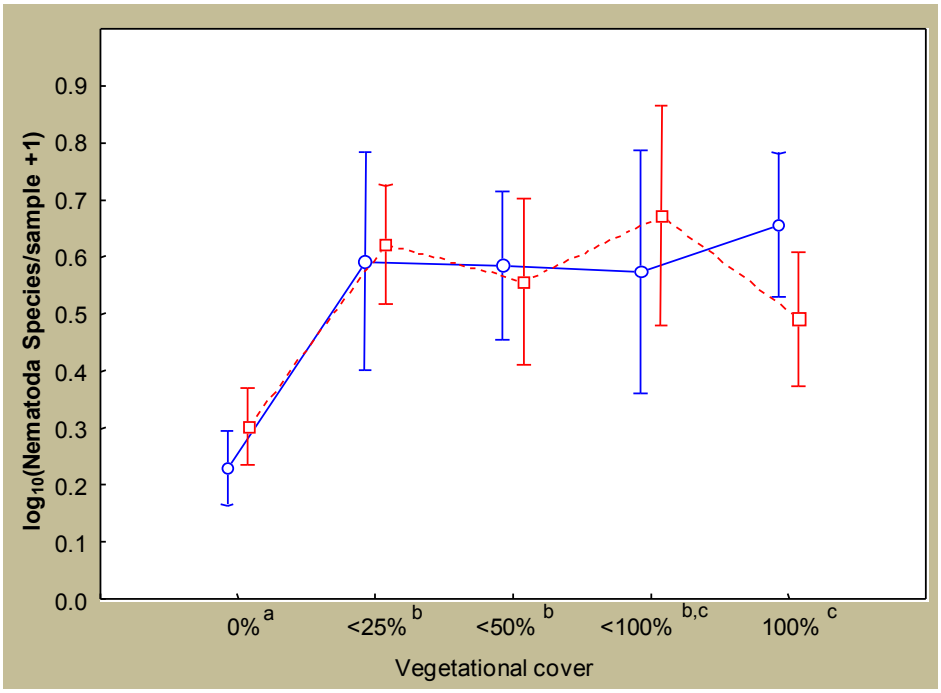


Fig. 161: Number of nematode species per sample in 2011 for anthropogenically influenced (blue) and non-influenced (red) areas in relation to vegetational cover. Figure explanation as in Fig. 160.

A strong correlation existed between locality (longitude and latitude) and nematode individual densities as well as species richness (Appendix 4, Table A4-5). The nematode communities became poorer from northeast to southwest, in respect to both densities as well as species richness. The densities and species numbers of the Antarctic nematode fauna correlated furthermore in both study years clearly with vegetational development - based upon percent

vegetational cover - and soil moisture (Appendix 4, Table A4-5): the denser the vegetation and the moister the study site, the richer was the nematode fauna. Not as consistent in both study years, but nonetheless at times positively correlated were nematode numbers (individuals and in 2010 also species) and soil organic matter, measured as mass loss on ignition, nitrogen content, carbon content and C/N ratio (Appendix 4, Table A4-5). The correlation analyses furthermore showed a slight, if not always consistent relationship between soil particle size and densities as well as species numbers, with richer nematode communities in soils with higher contents of fine sand, clay or silt.

Very little is known about the feeding strategies of Antarctic nematodes species. Based upon the morphology of the mouth cavities and partly also mouthparts ("stylets", teeth and "spears"), the nutritional resources can be roughly estimated, especially when explicit nutrient-resource studies exist for related species of the same genus. In the present study the nematode communities were dominated by bacterial feeders (Fig. 162). In the study sites on King George Island and Ardley Island, relatively large, omnivorous species also occurred. These species fed predominantly on green algae, as could be determined by the typical intestinal colour. Particularly diverse feeding types were present in the nematode communities from King George Island. Most likely due to the rich vegetation present in these sites. Besides bacterial and algal feeders also fungivore-radicivore nematodes were found here, which pierce and suck roots and fungal hyphae. Carnivorous feeding types showed a very site-specific distribution: carnivorous nematodes, whose prey generally consists of protozoa and smaller soil animals (Nematoda, Tardigrada, Rotatoria), were recorded in the present study from Halfmoon Island (in both study years) and Hannah Point, where they accounted for a considerable part of the total nematode numbers. Furthermore, predatory nematodes occurred in smaller numbers in the sites near Arctowski Station and in the Biologenbucht on King George Island.

A constant anthropogenic effect on the abundance of different feeding types was evident especially for the bacterivore guild. Human influence thereby negatively affected the numbers of those nematode species that generally require bacteria as a food resource, although this could not be statistically ascertained in all study years (Fig. 162; Appendix 5, Table A5-7). In 2011 human influence also negatively affected nematodes requiring algae as a food or can be considered to be omnivorous (algivore-omnivore guild). However, no anthropogenic effect on this guild could be determined in 2010 or when both years were analyzed together.

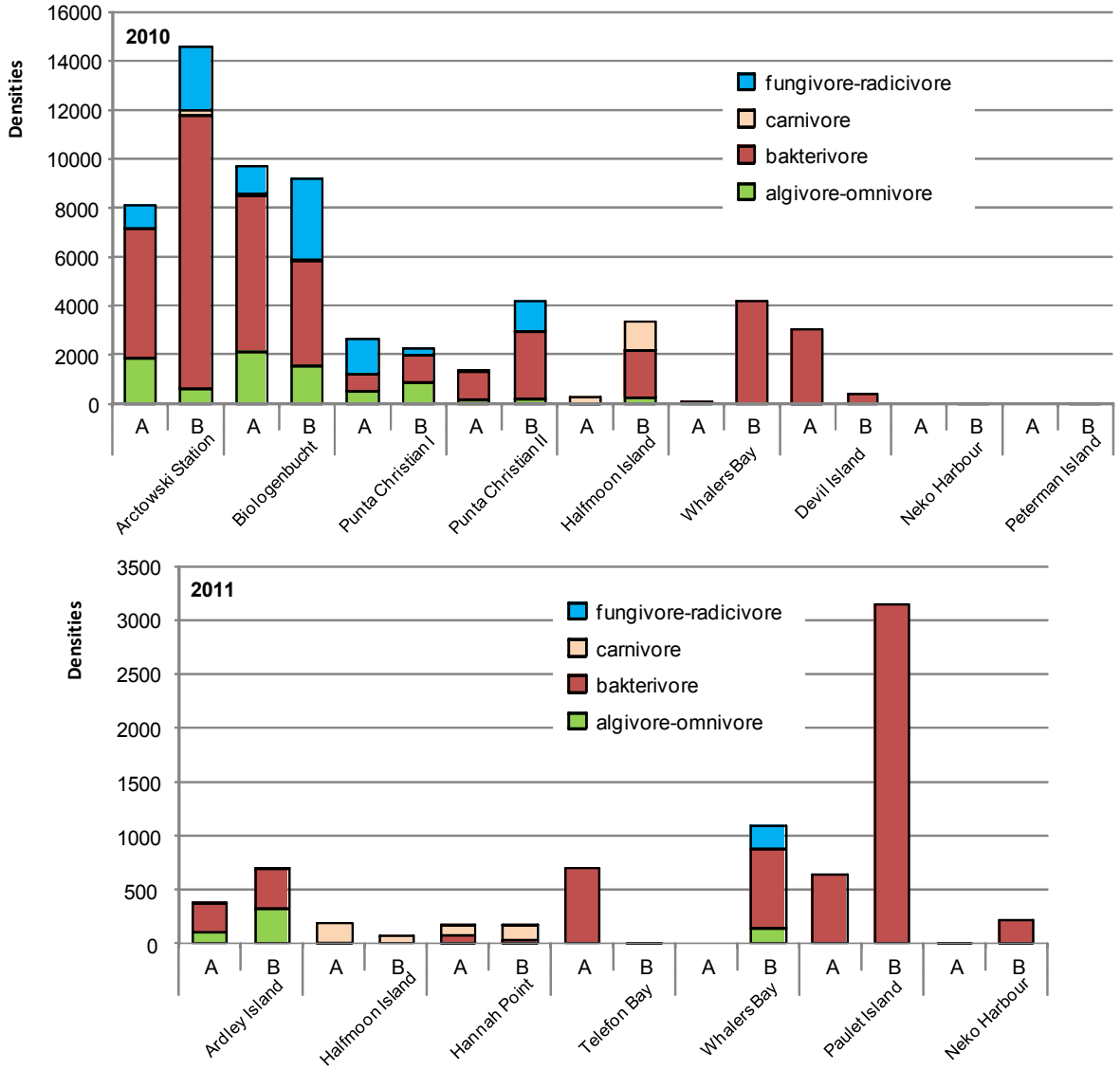


Fig. 162: Average densities (individuals per 100 cm^3 substrate) of the various feeding types recorded in 2010 (above) and 2011 (below) in the different localities, separate for anthropogenically influenced (A) and non-influenced (B) areas. The assignment of the recorded species to specific feeding types is given in Appendix 3, Table A3-4. The localities are sorted from left to right by increasing southern latitude. Please note the different scales of the y-axes.

3.5.1.2 Results and description of the recorded species

A total of 40 nematode species were recorded in the 13 study sites (localities) in the study years 2010 und 2011. The determined species are given in their systematic position in the following list. Nematode systematics follow Edaphobase (www.edaphobase.org) and Fauna Europea (www.faunaeur.org). The average densities of the individual species in the different localities are given in Appendix 3, Table A3-4.

Secernentea

Rhabditida

Rhabditoidea sp. 1

Cephalobidae

Acrobeloides arctowskii Holovachov & Boström, 2006
Cervidellus cf. vexilliger (de Man, 1880)
Heterocephalobus sp.

Panagrolaimidae
Panagrolaimus cf. *magnivulvatus* Boström, 1995
Diploscapteridae
Diploscapter sp.
Rhabditidae
Pellioditis cf. *marina* (Bastian, 1865)
Pelodera cf. *strongyloides* (Schneider, 1860)
Rhomborhabditis cf. *parateres* (Cobb, 1924)
Rhomborhabditis cf. *teres* (Schneider, 1866)

Aphelenchida
Aphelenchoididae
Aphelenchoides haguei Maslen, 1979
Aphelenchoides helicosoma Maslen, 1979
Aphelenchoides sp. 1

Tylenchida
Anguinidae
Ditylenchus parcevivens Andrássy, 1998
Tylenchidae
Filenchus sp. 1
Filenchus sp. 2

Adenophorea

Dorylaimida
Dorylaimida sp. 1
Dorylaimida sp. 2
Aporcelaimidae
Aporcelaimellus cf. *obtusicaudatus* (Bastian, 1865)
Dorylaimidae
Mesodorylaimus antarcticus Nedelchev & Peneva, 2000
Mesodorylaimus chipevi Nedelchev & Peneva, 2000
Mesodorylaimus sp. 1
Mesodorylaimus sp. 2
Nordiidae
Enchodelus signyensis Loof, 1975
Qudsianematidae
Amblydorylaimus isokaryon (Loof, 1975)
Eudorylaimus coniceps Loof, 1975
Eudorylaimus pseudocarteri Loof, 1975

Monhysterida
Monhysteridae
Eumonhystera vulgaris (de Man, 1880)
Eumonhystera sp. 1
Geomonhystera villosa (Bütschli, 1873)

Mononchida
Mononchidae
Coomansus gerlachei (de Man, 1904)

Plectida
Plectidae
Ceratoplectus armatus (Bütschli, 1873)
Plectus antarcticus de Man, 1904
Plectus belgicae de Man, 1904
Plectus insolens Andrássy, 1998
Plectus tolerans Andrássy, 1998
Plectus sp. 1

Teratocephalidae
Teratocephalus rugosus Maslen, 1979
Teratocephalus tilbrooki Maslen, 1979

Triplonchida
Prismatolaimidae
Prismatolaimus sp.

From these species, 21 were previously known from the Antarctic (Andrássy 1998, Nedelchev & Peneva 2000, Holovachov & Boström 2006). Four of these 21 Antarctic species exhibit a broad (global) distribution (Andrassy, 1998): *Ceratoplectus armatus*, *Coomansus gerlachei*, *Eumonhystera vulgaris* and *Geomonhystera villosa*. The remaining 17 species are, however, only known from the Antarctic and therefore can be considered to be endemic: *Acrobeloides arctowskii, Amlydorylaimus isokaryon, Aphelenchoides haguei, A. helicosoma, Ditylenchus parcevivens, Enchodelus signyensis, Eudorylaimus coniceps, E. pseudocarteri, Mesodorylaimus anarcticus, M. chipevi, Panagrolaimus* cf. *magnivulvatus, Plectus antarcticus, P. belgicae, P. insolens, P. tolerans, Teratocphalus rugosus, T. tilbrooki*.

Aporcelaimellus cf. *obtusicaudatus, Cervidellus* cf. *vexilliger* as well as the Rhabditidae *Pelodera* cf. *strongyloides, Pellioditis* cf. *marina* and *Rhomborhabditis* cf. *teres* are recorded from the Antarctic for the first time in the present study, although Andrássy (1998) states that the genera *Cervidellus* and *Pelodera* have been previously mentioned for the maritime Antarctic nematode fauna, albeit without specification of the specific species. A further 13 species could be putatively but clearly discerned when mounted in the microscopic slides, but could not yet be determined to species level (Appendix 3, Table A3-4), because they have either not yet been described (= are new to science) or while the necessary determination literature was not available. These taxa possibly represent South African or South American species.

With the relatively detailed investigation of the 13 different Antarctic study sites (localities), the present investigations contribute vastly to the assessment of the nematode fauna of Maritime Antarctica (Fig. 163). With the exception of Deception Island, Livingston Island and Arctowski Station on King George Island, the localities studied here - which were chosen primarily due to their touristic and scientific use – were to the best of our knowledge for the first time the subject of extensive nematological investigations.

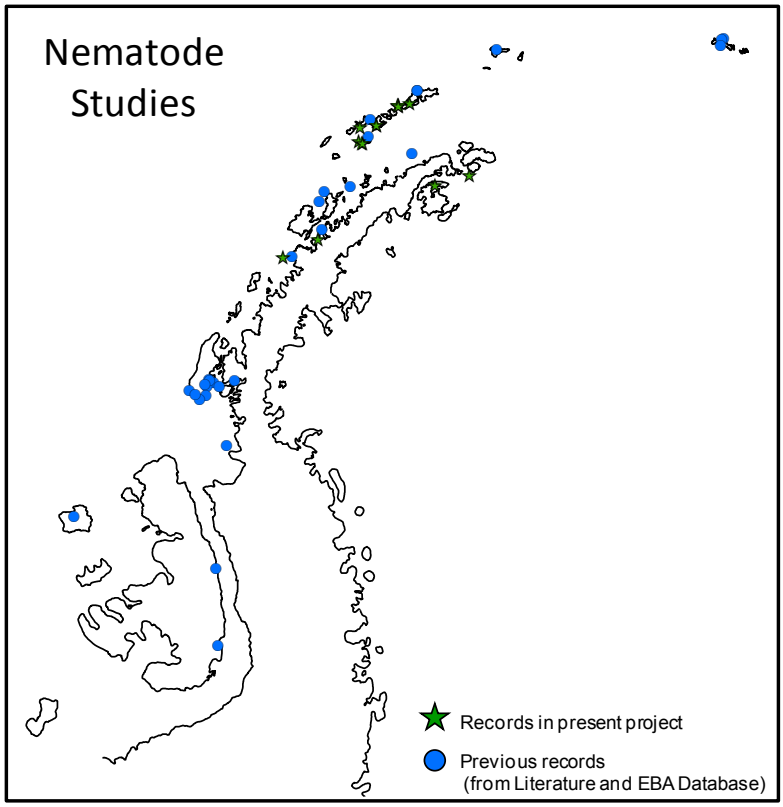


Fig. 163: Previous and present records of the nematode species determined in the present study.

As with the total densities and species richness, also the abundances of most of the individual species were dependent upon locality (longitude and latitude; Appendix 4, Table A4-5). The populations became individual poorer from northeast to southwest. Individual numbers of many species correlated furthermore in both years positively with the degree of vegetation, measured as percent vegetational cover, and negatively with soil temperature. Not as consistent, but when significant then always with a positive correlation, a relationship between the abundances of a number of species and soil organic matter (measured as mass loss at ignition) could be determined (Appendix 4, Table A4-5).

Human trampling measurably affected the abundances of only a few species (Appendix 5, Table A5-7). For some of these species, opposite anthropogenic effects could be ascertained in 2010 and 2011; human influence was in the one year apparently beneficial, in the other however more detrimental. Four species, for which a consistent anthropogenic impact could be determined in both years, showed negative effects: the algivore-omnivore species *E. coniceps* and *E. pseudocarteri* as well as the bacterivore species *P.* cf. *magnivulvatus* and *R.* cf. *parateres* exhibited higher abundances in the anthropogenically non-influenced areas (Appendix 5, Table A5-7).

The ANCOVAs also showed an anthropogenic impact for some species. However, since the influence of vegetation and other habitat parameters were filtered out during these analyses, the affected species were not necessarily the same as those for which an impact could be shown with the ANOVAs. More frequently, however, the ANCOVAs revealed the influence of vegetational cover on the abundances of individual species. The degree of vegetational cover showed different effects depending on whether an anthropogenic influence was present or not,

as is apparent in the statistically significant interaction between human influence and educational cover (Appendix 6, Table A6-5, "Anthrop. x Vegetation").

3.5.1.2.1 Indigenous Antarctic species (selection)

The specific ecological characteristics and the geographical distribution of six of the total 40 nematode species are described in more detail in the following. These six species were selected so that each of the different feeding types is represented by at least one species. Also, both endemic species as well as those with a broader distribution are presented and the species specificity of anthropogenic impacts as well as relationships between species' distributions and specific environmental parameters are shown in the examples presented here.

Coomansus gerlachei (de Man, 1904)

C. gerlachei occurs in regions of high northern or southern latitudes, for instance also in Canada, and is widely distributed throughout the Antarctic islands (Spaull 1973, Maslen 1979a, 1979b). Peneva et al. (1996) report records on Livingston Island (Fig. 164). In the present study, individuals of the species were found in high frequencies (e.g., on Halfmoon Island in 100% (2010) and 92% (2011) of the samples) and surprisingly high densities (on average 113 – 555 individuals per 100 cm^3 substrate; Appendix 3, Table A3-4) on Halfmoon Island (in both years) and Hannah Point. *C. gerlachei* is a relatively large, predatory nematode species. It was the only nematode species in the present study exhibiting a significant positive correlation to soil nitrogen content. Its high dominance within the nematode communities of the locations mentioned above - in 22 of 36 studied samples from Halfmoon Island, *C. gerlachei* was the only recorded nematode species – indicates that it does not primarily feed from other nematodes species, but probably from juvenile individuals of its own species as well as Rotatoria, Protozoa and smaller Tardigrada.

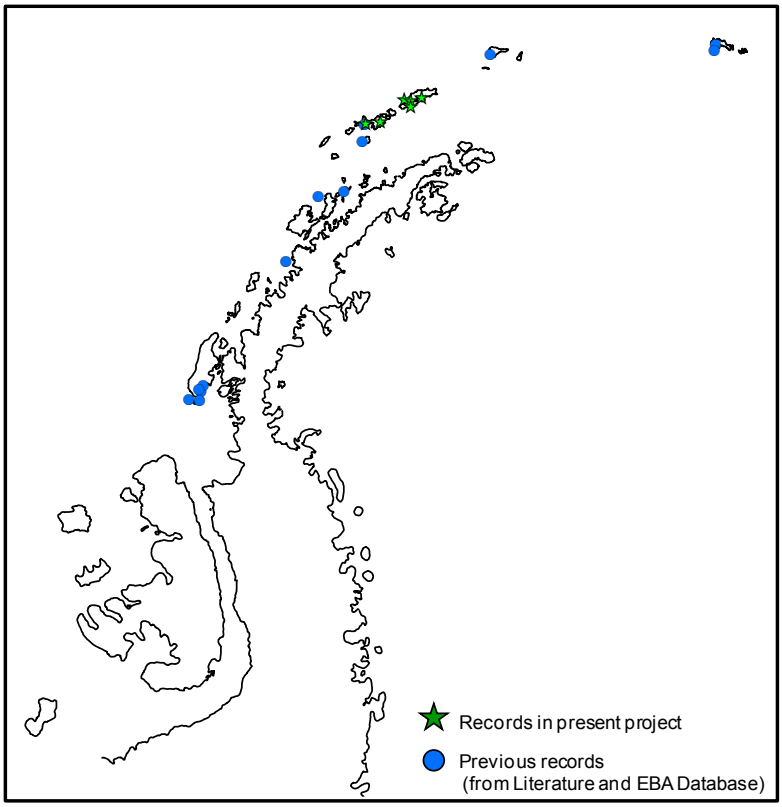


Fig. 164: Records of the species *Coomansus gerlachei* in the maritime Antarctic mentioned in the literature and/or the EBA-Datenbank (blue dots) as well as determined in the present study (green stars).

Panagrolaimus cf. *magnivulvatus* Boström, 1995

Panagrolaimus magnivulvatus has to date only been reported from Continental Antarctica. The species was first described by Boström (1995) from nunataks in Dronning Maud Land, eastern Antarctica. To our knowledge, the records of *Panagrolaimus* cf. *magnivulvatus* in the present study from Arctowski Station, Halfmoon Island, Hannah Point, Telefon Bay, Whalers Bay, Paulet Island, Devil Island and Neko Harbour are the first from Maritime Antarctica (Appendix 3, Table A3-4). A confirmation of the determination is still pending. Further, but not more specifically identified records of *Panagrolaimus* are known from the Antarctic islands (Fig. 165). Whether these concern the same species must also still be clarified.

P. cf. *magnivulvatus* belongs to one of the few nematode species which exhibited a significant human impact in the present study. Its populations were reduced in anthropogenically influenced areas (Fig. 166; Appendix 5, Table A5-7).

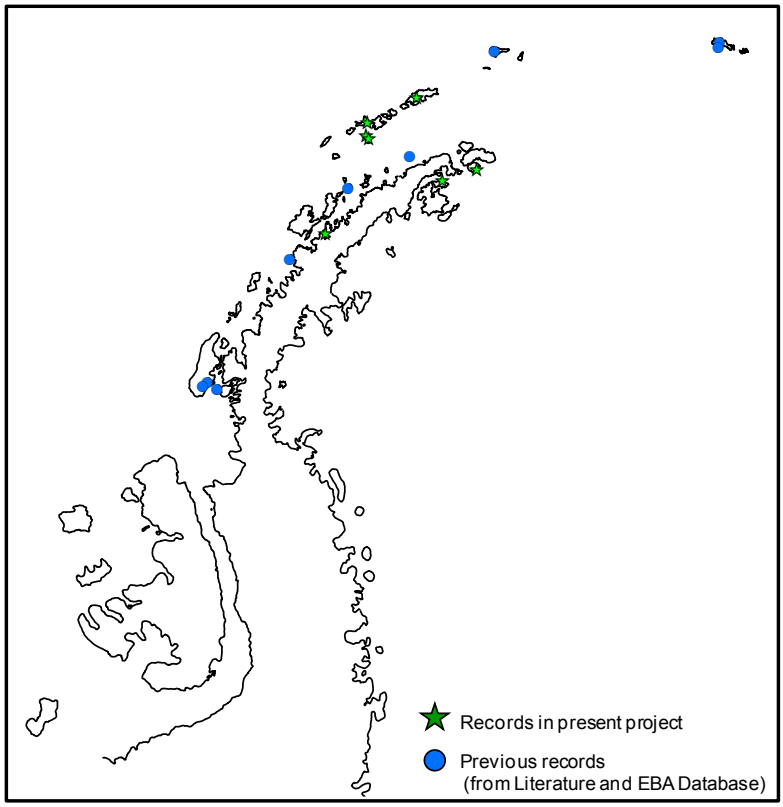


Fig. 165: Records of the genus *Panagrolaimus* in the maritime Antarctic mentioned in the literature and/or the EBA-Datenbank (blue dots) as well as of *Panagrolaimus* cf. *magnivulvatus* determined in the present study (green stars).

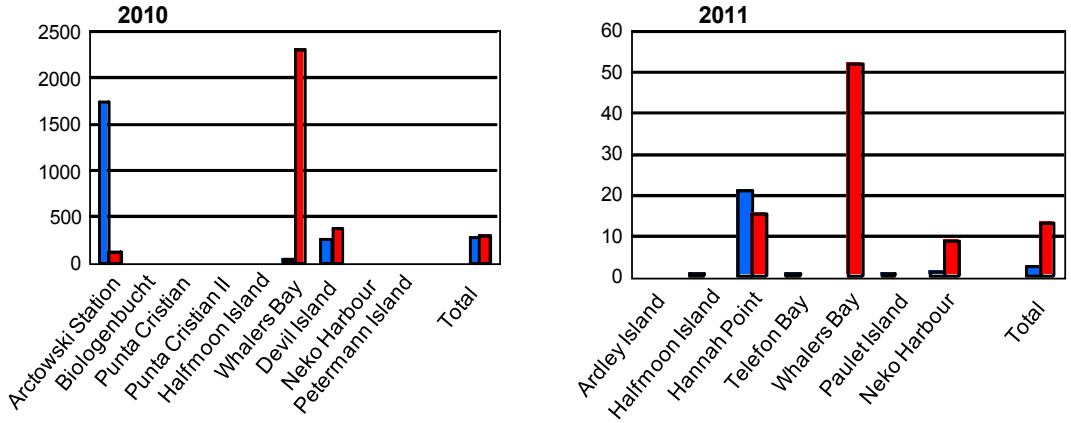


Fig. 166: Densities of *Panagrolaimus* cf. *magnivulvatus* (individuals per 100 cm^3 substrate) in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 and 2011.

Rhomborhabditis cf. *teres* (Schneider, 1866)

The ornithogenic soils on Paulet Island represented the habitat for *R.* cf. *teres*, a species which – with an average of 2309 individuals per 100 cm^3 substrate - predominated in all of the 12 samples taken from this island (Appendix 3, Table A3-4). This species, which morphologically strongly resembles *R. teres*, is reported here for the first time from Antarctica. A confirmation of its identification is still pending. Representatives of *R. teres* feed on bacteria and reproduce explosively where fresh carbon- and nitrogen-rich sources ensure a strong bacterial growth, e.g. on carrion or dung. On Paulet Island, these nutrients originate almost exclusively from bird droppings (guano). With an average of 1.53% (at 19.7% organic material), the nitrogen contents

of these soils were higher than in all other localities of the present study (average nitrogen content of all locations: 0.33%; organic material: 5.5%). In the samples from 2011, a corresponding strong positive relationship between individual numbers of *R.* cf. *teres* and soil nutrient content (measured as soil organic material content [mass loss at ignition], soil nitrogen and carbon content) could be determined (Appendix 4, Table A4-5).

Aphelenchoides haguei Maslen, 1979

A. haguei was isolated and described for the first time by Maslen (1979a) from moss patches on Signy Island. Since then the species has also been recorded from soil samples and thus exhibits a broad distribution throughout the maritime Antarctic (Fig. 167). *Aphelenchoides* species possess a stylet in their oral cavity, with which they pierce plant-root and fungal cells and suck out their contents. The majority of *Aphelenchoides* species, for which the nutrient-resource spectrum has been studied, feed primarily on fungi. Some species can also survive and reproduce after feeding on algae, lichens, root epidermis cells or hair-root cells (Yeates et. al 1993). Due to the almost totally lacking vegetation in the study sites of Whalers Bay on Deception Island (vegetational cover 0% or sporadically < 25 %), it can be assumed that representatives of *A. haguei* feed here exclusively from fungal cells and also possibly algae. On King George Island, plant root cells can also be an additional food resource.

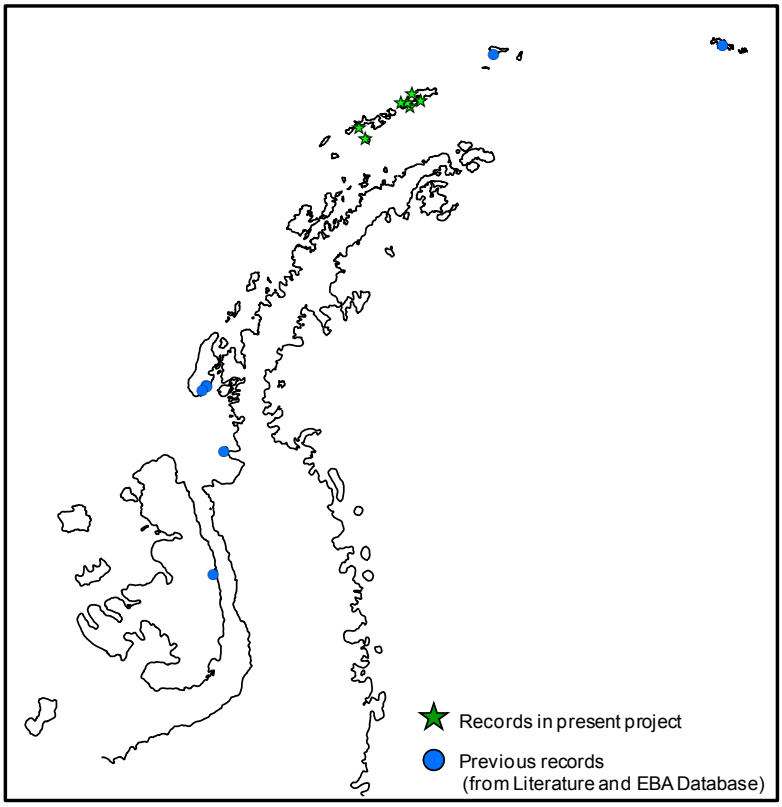


Fig. 167: Records of *Aphelenchoides haguei* in the maritime Antarctic mentioned in the literature and/or the EBA-Datenbank (blue dots) as well as determined in the present study (green stars).

Acrobeloides arctowskii Holovachov & Boström, 2006

The very distinctive nematodes species *A. arctowskii* had previously been recorded exclusively from King George Island, from where it was originally described by Holovachov & Boström (2006) from the vicinity of the Polish Antarctic station „Henryk Arctowski“ (Fig. 168). In the present study, this site of occurrence could be confirmed. As in the original description, *A.*

arctowskii was determined from samples near Arctowski Station, in which the higher plant species *Deschampsia antarctica* rooted. The present study could furthermore increase the distributional area of the species by the addition of Deception Island, where it was recorded from Whalers Bay in both study years (2010 and 2011). Since the corresponding samples from these sites were devoid of vegetation (with the exception of the sporadically occurring algae *Prasiola crispa*), the records of this species suggest that an association with *Deschampsia antarctica* or other higher plants is not a prerequisite for a colonization success by *A. arctowskii*. Based upon the mouthparts in its oral cavity, *A. arctowskii* is likely a bacterial feeder and possibly also facultatively fungivorous, as are other species of the genus *Acrobeloides*.

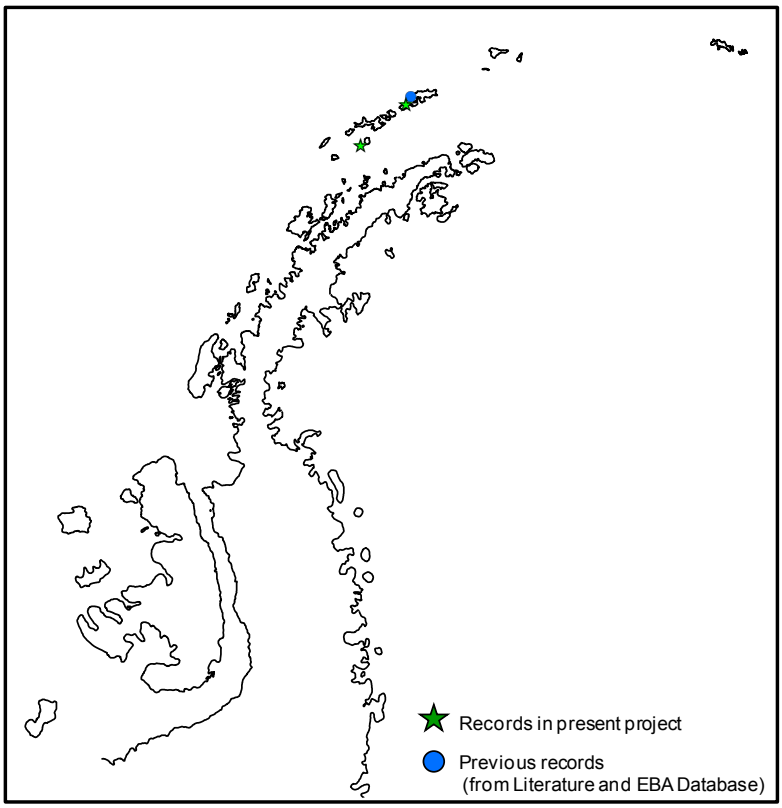


Fig. 168: Records of *Acrobeloides arctowskii* in the maritime Antarctic mentioned in the literature and/or the EBA-Datenbank (blue dots) as well as determined in the present study (green stars).

<u>*Eudorylaimus coniceps* Loof, 1975</u>

E. coniceps is widely distributed throughout the maritime Antarctic. Previous records extend from Signy Island in the north to Alexander Island in the south of the Antarctic Peninsula (Andrássy 1998 and EBA-Datenbank; Fig. 169). The present investigation could expand records of this species to King George Island and Livingston Island. *E. coniceps* is among one of the few nematode species in the present study for which a significant human impact could be recognized. Its populations were significantly reduced in areas of anthropogenic influence (Fig. 170; Appendix 5, Table A5-7).

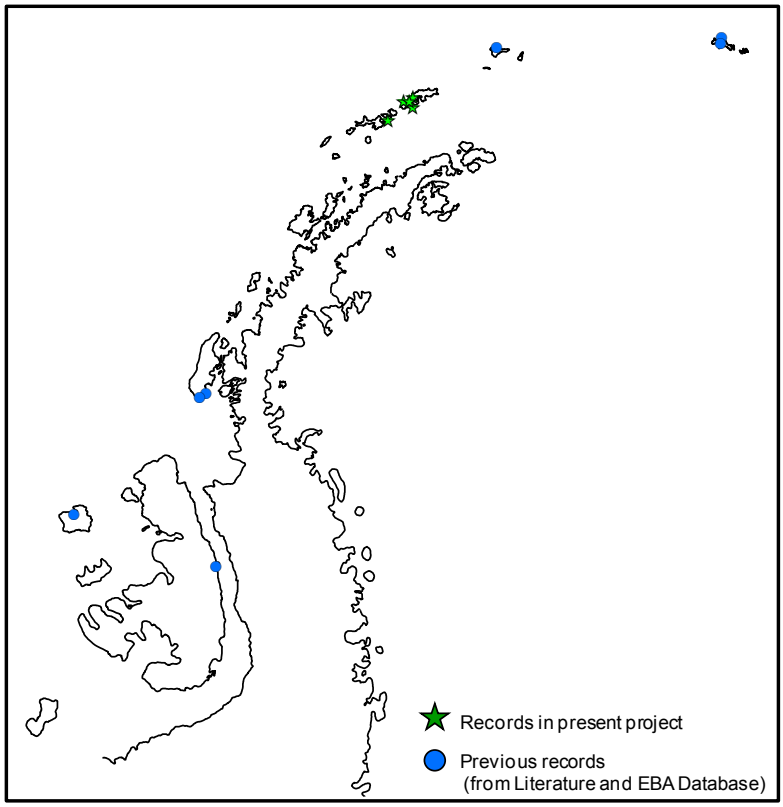


Fig. 169: Records of *Eudorylaimus coniceps* in the maritime Antarctic mentioned in the literature and/or the EBA-Datenbank (blue dots) as well as determined in the present study (green stars).

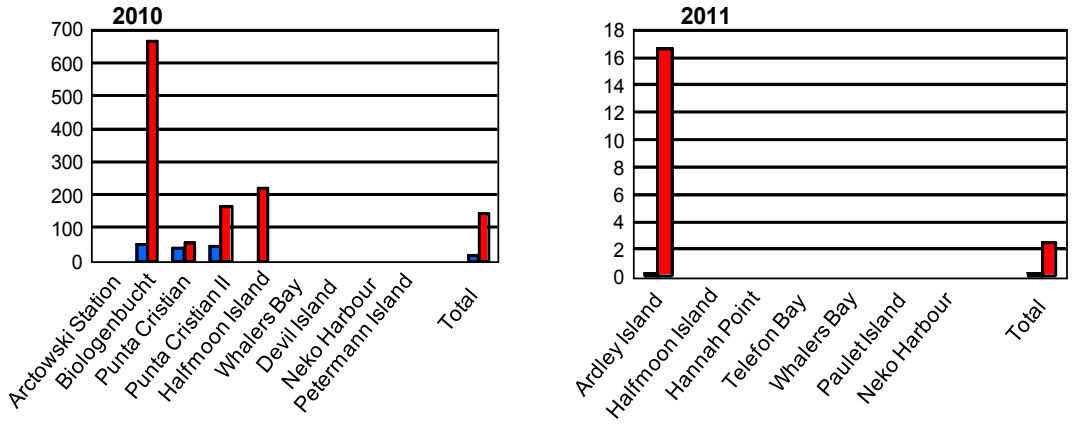


Fig. 170: Densities of *Eudorylaimus coniceps* (individuals per 100 cm^3 substrate) in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 and 2011.

3.5.1.2.2 Potentially introduced (non-native) species

No non-native species of the Nematoda could be determined with certainty in the present study.

3.5.2 Tardigrada (Water Bears)

Water Bears, belonging to the phylum Tardigrada, are small metazoans ranging in size from 50 to 1200 µm that occur worldwide in marine, freshwater and terrestrial habitats. In terrestrial habitats tardigrades occur wherever free water is available, such as cushion plants, mosses, lichens, leaf-litter and in the pore spaces of soils. Although small, tardigrades have a complete digestive system and feeding strategies range from grazing bacteria, feeding on individual cells of algae, moss and lichens, to predation on other meso- and microfauna (tardigrades, nematodes, rotifers).

The earliest record for Antarctic tardigrades dates to Richters (1904), who reported *Acutuncus antarcticus* (originally described as *Macrobiotus antarcticus*) from Gaussberg (66° 50'S, 89° 11'E). Two years later Murray (1906) described three new species of tardigrades, with a number of unidentified species, from Laurie Island, South Orkney Islands (60° 43' 59"S, 44° 36' 58"W). Approximately 70 species have now been described from Antarctica, with new additions occurring regularly (Convey & McInnes 2005, McInnes, personal database). Minimal predation pressure and a perennial abundance of food resources allow population densities of Antarctic tardigrades in some habitats to exceed those of temperate or tropical regions 10- to 1000-fold (Jennings 1979; McInnes & Pugh 1999). The tardigrade fauna is therefore an important component of the Antarctic terrestrial biomass (Convey & McInnes 2005).

3.5.2.1 General faunistical parameters

Not only the distribution of individual species (see below), but also community-ecological parameters reveal the very site-specific characteristic of Antarctic tardigrade coenoses. For instance, statistically significant differences between the studied localities in regards to both total densities as well as species richness could be discerned in both study years (Fig. 171; Appendix 5, Table A5-1).

A comparison of the tardigrade abundances in anthropogenically influenced and non-influenced areas indicate a significant human impact particularly in the data from 2010, with higher densities in the non-influenced areas (Fig. 172; Appendix 5, Table A5-8). The recorded tardigrade numbers were in total lower in 2011. Analyzing both study years together revealed overall lower individual numbers in the human influenced areas. In contrast, a statistically significant influence of human trampling on the number of recorded species could not be determined.

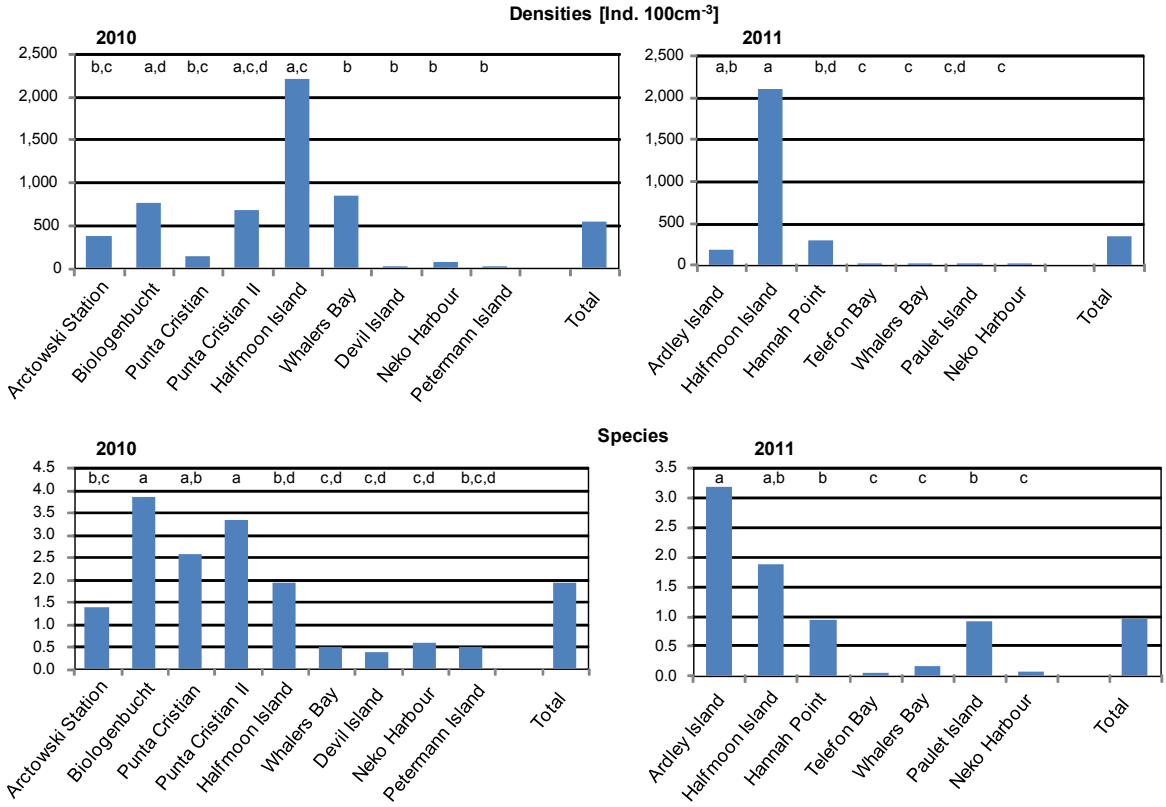


Fig. 171: Tardigrade densities (individuals per 100 cm^3 substrate) and species number per sample recorded in the various study sites in 2010 and 2011. Study sites are ordered from left to right by increasing southern latitude. Different letters denote significant differences between sites; the densities or species richness in sites with a common letter were statistically not different. Please note the different scales of the y-axes.

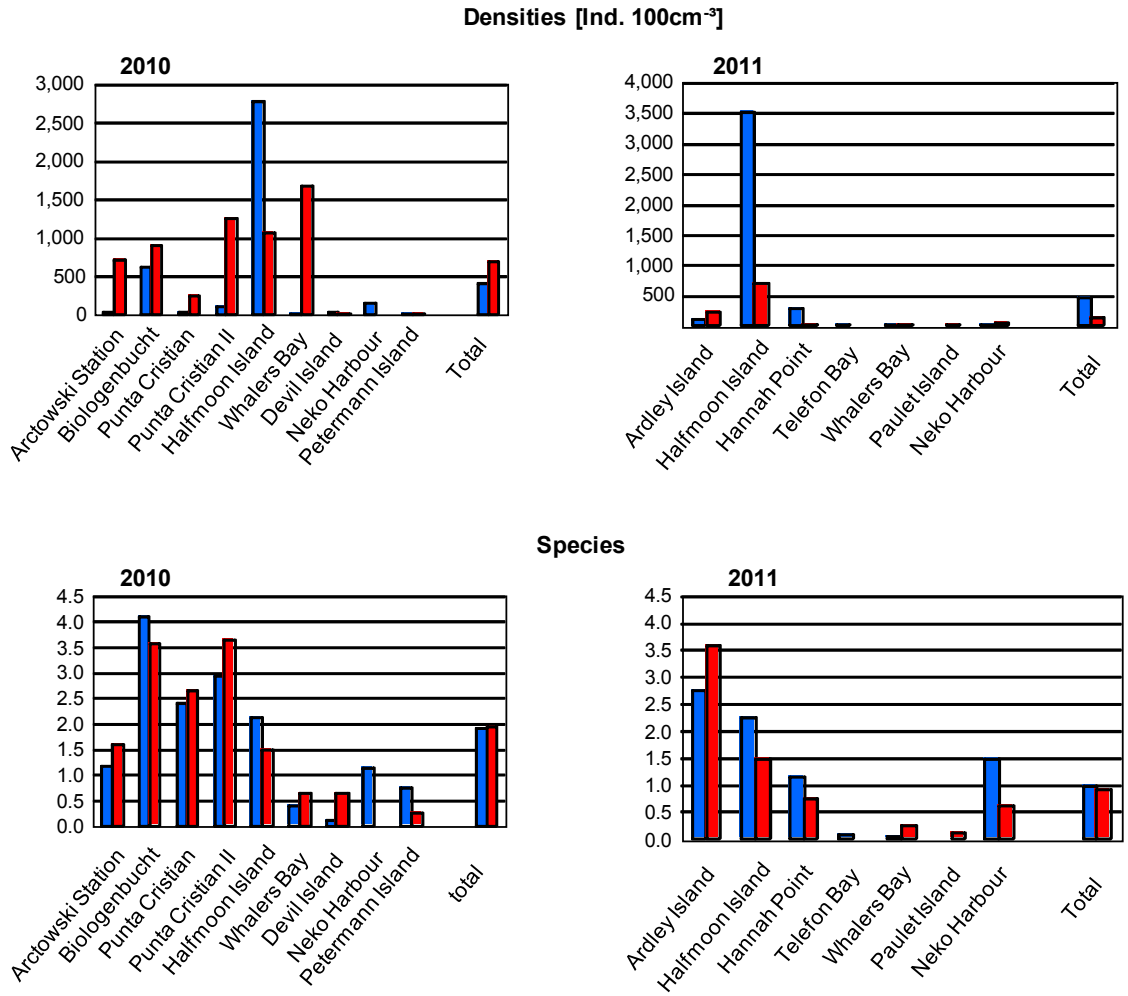


Fig. 172: Tardigrade densities (individuals per 100 cm^3 substrate) and number of species per sample in the various study sites, separately for anthropogenically influenced (blue) and non-influenced (red) areas, as well as for 2010 and 2011. Please note the different scales of the y-axes.

3.5.2.2 Results and description of the recorded species

The tardigrade species recorded in the present study are given in their systematic position in the following list. Tardigrade systematics follows the current checklist from 2012 by Degma, Guidetti & Bertolani (www.tardigrada.modena.unimo.it). Their average densities in the various localities are given in Appendix 3, Table A3-6.

Eutardigrada

Hypsibioidea

Calohypsibiidae

Calohypsibius Thulin, 1928 sp.
Hexapodibius Pilato, 1969 sp.

Hypsibiidae

Acutuncus antarcticus (Richters, 1904)
Diphascon Plate, 1889 sp. (*Adropion* + *Diphascon*)
Diphascon (*Adropion*) Pilato, 1987
Diphascon (*Diphascon*) Plate, 1889
Hypsibius cf. *dujardini* (Doyère, 1840)
Simplex moult Gen. sp. (possibly *Acutuncus, Diphascon* or *Hypsibius*)

Isohypsibioidea

Isohypsibiidae

Isohypsibius Thulin, 1928 sp. 1
Isohypsibius Thulin, 1928 sp. 2
Ramajendas cf. *frigida* Pilato & Binda, 1990

Macrobiotoidea
Macrobiotidae
Macrobiotus cf. *furciger* Murray, 1907
Murrayidae
Dactylobiotus R.O. Schuster, 1980 sp.

Heterotardigrada

Echiniscoidea
Echiniscidae
Echiniscus jenningsi Dastych, 1984
Echiniscus (= *Testechiniscus*) *meridionalis* Murray, 1906
Pseudechiniscus Kristensen, 1987 sp.

The determined species are briefly described in the following.

Ramajendas cf. *frigidus* Pilato & Binda, 1990

This species is currently known from Continental Antarctica (Victoria Land) and Maritime Antarctica (Antarctic Peninsula and islands up to the South Sandwich Islands). Little is known of its habitat preferences, but previous records have come from both semi-aquatic and drier terrestrial sites.

Acutuncus antarcticus (Richters, 1904)

A. antarcticus is an Antarctic species found on the continental and maritime Antarctic, including South Georgia. It is commonly found in semi-aquatic to aquatic conditions.

Macrobiotus cf. *furciger* Murray, 1907

At the moment the extracted samples were conserved, a number of females of this taxa carried developed eggs nearly ready for oviposition. These eggs allowed this taxon to be identified as *Macrobiotus* cf. *furciger*. The distributional range of the species includes the maritime Antarctic and South Georgia. *M. furciger* was recently diagnosed as a group of species and re-described. However, taxonomic problems regarding this species still exist, since the original type name has been assigned a new type locality (Locus typicus) and a new species allocated to the original type locality. This species-group is usually found in drier terrestrial sites.

Diphascon (*Adropion*) Pilato, 1987 spp (2 species)

Two species of this subgenus could be identified in individuals mounted on the permanent slides.

Diphascon (*Adropion*) *greveni* Dastych, 1984

The type locality of *D. greveni* is King George Island. The distribution of this species ranges throughout the maritime Antarctic and South Georgia. It is found in both semi-aquatic and drier terrestrial sites.

Diphascon (*Adropion*) Pilato, 1987 sp.

Diphascon (*Adropion*) sp. is possibly a new species, as the individuals of this taxon could not be determined to other species in the region with the current determination keys. Its habitat preferences have not yet been explored.

Diphascon (*Diphascon*) Plate, 1889 spp (3 species)

Three species of this subgenus were found in the samples.

Diphascon (*Diphascon*) *mirabilis* Dastych, 1984

The type locality of *D. mirabilis* is King George Island. The species' distribution probably ranges throughout the maritime Antarctic. Its habitat preferences are not clear; it has been found in drier terrestrial habitats, but may also inhabit semi-aquatic biotopes.

Diphascon (*Diphascon*) cf. *pingue* (Marcus, 1936)

D. cf. *pingue* has been recorded from maritime Antarctic sites as well as South Georgia. *D. pingue* sensu stricto is a species occurring in the northern hemisphere, but the taxon's name covers a species complex. The specimens from Antarctica require further investigation to identify the true species indentify and may include one or more new species. In the Antarctic, *D.* cf. *pingue* has been reported from a wide range of habitats from aquatic to drier terrestrial sites.

Diphascon (*Diphascon*) *langhovdensis* (Plate, 1888)

D. langhovdensis has been recorded from the continental and maritime Antarctic. There is little information about the habitat preferences of this taxon.

Hypsibius cf. *dujardini* (Doyère, 1840)

This species has been reported as *Hypsibius* cf. *dujardini* from the maritime Antarctic and South Georgia. Individuals of this taxon recorded in the current project represent an undescribed new species. The species is usually found in semi-aquatic habitats, slightly drier than those in which *Acutuncus antarcticus* have been found.

Isohypsibius Thulin, 1928 spp (2 species)

Isohypsibius improvises Dastych, 1984

Two specimens were detected that could be tentatively identified as *Isohypsibius improvises*. This species is known from Enderby Land and King George Island. These two specimens do not show the characteristic morphological "cavities" in the cuticle and therefore may represent a new species, although too few specimens were available to allow a taxonomic analysis. Its habitat preferences are unknown.

Isohypsibius Thulin, 1928 sp.

A second *Isohypsibius* sp. was found that possessed cuticular gibbosities, which is characteristic for a number of species of the genus *Isohypsibius*. This type of *Isohypsibius* has not previously been observed in the Antarctic. The individuals therefore possibly represent a new species, but this needs further taxonomic analysis, which would require more specimens. The habitat preferences of this taxon are based on the current study: relatively dry soils associated with vegetation.

Dactylobiotus R.O. Schuster, 1980 sp.

No eggs were present in the few specimens of this taxon collected in the current study. As eggs are necessary for taxonomic determination, it was therefore not possible to identify the specimens to a species level. They are most likely similar to specimens previously reported from the maritime Antarctic as *Dactylobiotus* cf. *ambiguus,* but this requires further taxonomic

analysis to rename the taxon as a new species. Previously, it was usually present only in aquatic habitats.

Echiniscus jenningsi Dastych, 1984

The type locality of *E. jenningsi* is King George Island and the species is present throughout the maritime Antarctic and South Georgia. Its habitat preference can range from marginal aquatic to drier terrestrial sites.

Echiniscus meridionalis Murray, 1906

The type locality of this taxon is Laurie Island (South Orkney Islands) and it has also been reported from King George Island. The species is usually found in drier terrestrial habitats.

Pseudechiniscus Kristensen, 1987 sp.

This taxon has been previously reported as *Pseudechiniscus* cf *suillus* from the continental and maritime Antarctic as well as from South Georgia. These and the current records represent an as of yet undescribed new species found in drier terrestrial habitats.

Hexapodibius boothi Dastych & McInnes, 1994

Dundee Island is the type locality for this species and an unidentified *Hexapodibius* species has also been reported from King George Island. The species is present in soils.

Calohypsibius Thulin, 1928 sp.

This species has been previously reported as *Calohypsibius* cf. *ornatus* (representing a new species) from a number of Subantarctic islands, including South Georgia, but had not previously been recorded from the maritime Antarctic. Mosses and lichens from relatively dry terrestrial sites have been reported as habitat preferences.

A number of individuals of the genera *Hypsibius*/*Diphascon*/*Acutuncus* were found in the morphological simplex form (without mouthparts). Since the mouthparts are necessary for determination, the animals were unidentifiable to species level.

At the species level, some statistically significant results were obtained concerning an anthropogenic influence on populations. The relatively rare species *Echiniscus meridionalis* showed consistent results over the two year period, occurring more strongly in non-influenced areas (Fig. 173; Appendix 5, Table A5-8). The combined group of *Diphascon* species, which occurred in a slightly broader range of localities, was also found in higher densities in non-influenced areas (Fig. 174).

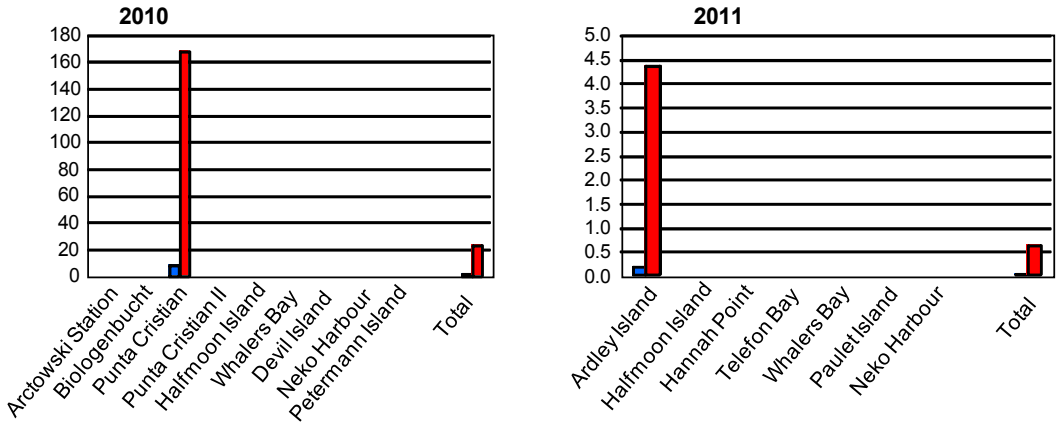


Fig. 173: Densities of *Echiniscus meridionalis* (individuals per 100 cm^3 substrate) in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 and 2011.

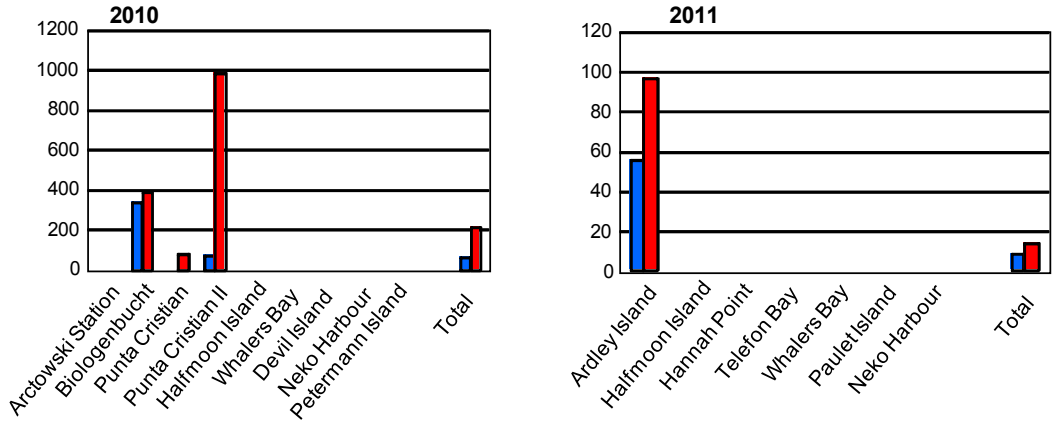


Fig. 174: Densities of *Diphascon* (*Diphascon*) and *Diphascon* (*Acropion*) species (individuals per 100 cm^3 substrate) in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 and 2011.

Species which covered the full sampling range, such as *Acutuncus antarcticus*, *Macrobiotus* cf. *furciger* and *Hypsibius* cf. *dujardini*, suggest more mixed signals of tolerance and/or of being affected by anthropogenic influence (Fig. 175; Appendix 5, Table A5-8). When analyzing both study years together, only *Macrobiotus* cf. *furciger* showed significantly lower densities in the anthropogenically influenced areas.

Although many of the species showed statistically significant results regarding human impact (Appendix 5, Table A5-8), their population numbers were so low as to make these results almost meaningless. However, such low population numbers are susceptible to any changes in habitat conditions, so that a human impact on these species could indeed be very significant.

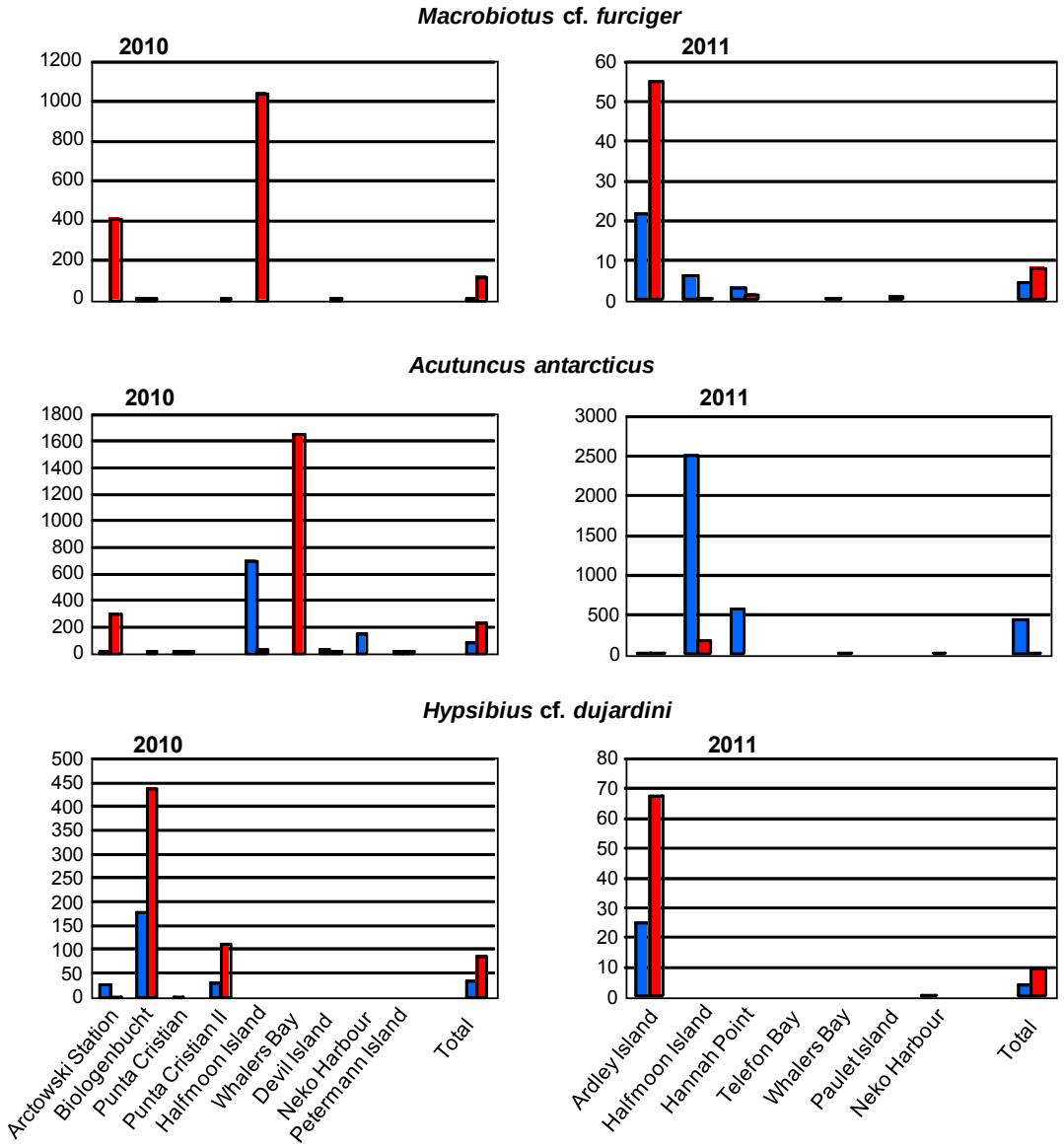


Fig. 175: Densities of *Macrobiotus* cf. *furciger*, *Acutuncus antarcticus* and *Hypsibius* cf. *dujardini* (all in individuals per 100 cm^3 substrate) in the anthropogenically influenced (blue) and non-influenced (red) areas in 2010 and 2011.

Species such as *Echiniscus meridionalis* were not only restricted to certain localities, but also have specific habitat preferences. Thus their close association with vegetation is not unexpected (Appendix 6, Table A6-6). An anthropogenic influence was hereby often stronger (or even only detectable) at higher levels of vegetational cover (Appendix 6, Table A6-6, interaction "Anthrop. x Vegetation"). As long as tourists etc. avoid trampling obvious vegetation, this species will be preferentially found in undisturbed habitats with moss growth. Species with a wider tolerance of habitat conditions, such as *Acutuncus antarcticus*, are less affected and may be tolerant of anthropogenic impact, particularly if the habitat conditions change in favour of the species' requirements. Obligate soil dwellers, e.g. *Hexapodibius* sp., are particularly vulnerable to human impact. However, the results obtained here are based on very small population numbers and the limited knowledge of this species' requirements precludes meaningful conclusions as to how tourism impacts this particular species.

3.5.2.3 Potentially introduced (non-native) species

The results of the present soil-zoological survey from human impacted regions of the maritime Antarctic do not indicate “introduced” tardigrade species. The species recorded in the present investigations have all previously been found in the maritime Antarctic and the Subantarctic South Georgia, with the exception of those species identified as potentially new to science.

3.6 Faunistical community-level (multivariate) data analyses

3.6.1 Collembola

A *non metric multidimensional scaling* ordination (NMDS) of all collembolan data showed a very tight cluster of points (overlapping dots to the right of the ordination center in Fig. 176, top), which were formed by a majority of the study areas (sampling plots). A group of further study areas were arranged in the periphery of the ordination, the collembolan communities of which obviously differed strongly from those within the point cluster. The areas arranged in the periphery of the ordination contained no or only very few animals and therefore were represented in the ordination displaced from the more individual-rich areas.

In order to better display this tight cluster, the NMDS was repeated using only these study areas. This showed a very strong influence of the factor “locality”: the areas of a locality almost always ordinated very closely to each other (Fig. 176, middle). If the points are displayed according to the factor “treatment” (human influence), then the ordination result showed no concrete pattern; the points are obviously randomly distributed (Fig. 176, bottom).

A PERMANOVA confirmed the NMDS: the factor “locality” had a very highly significant influence on the similarity of the Collembola communities. An interaction between the factors “locality” and “treatment” (= anthropogenic influence) was also not significant (Table 10.)

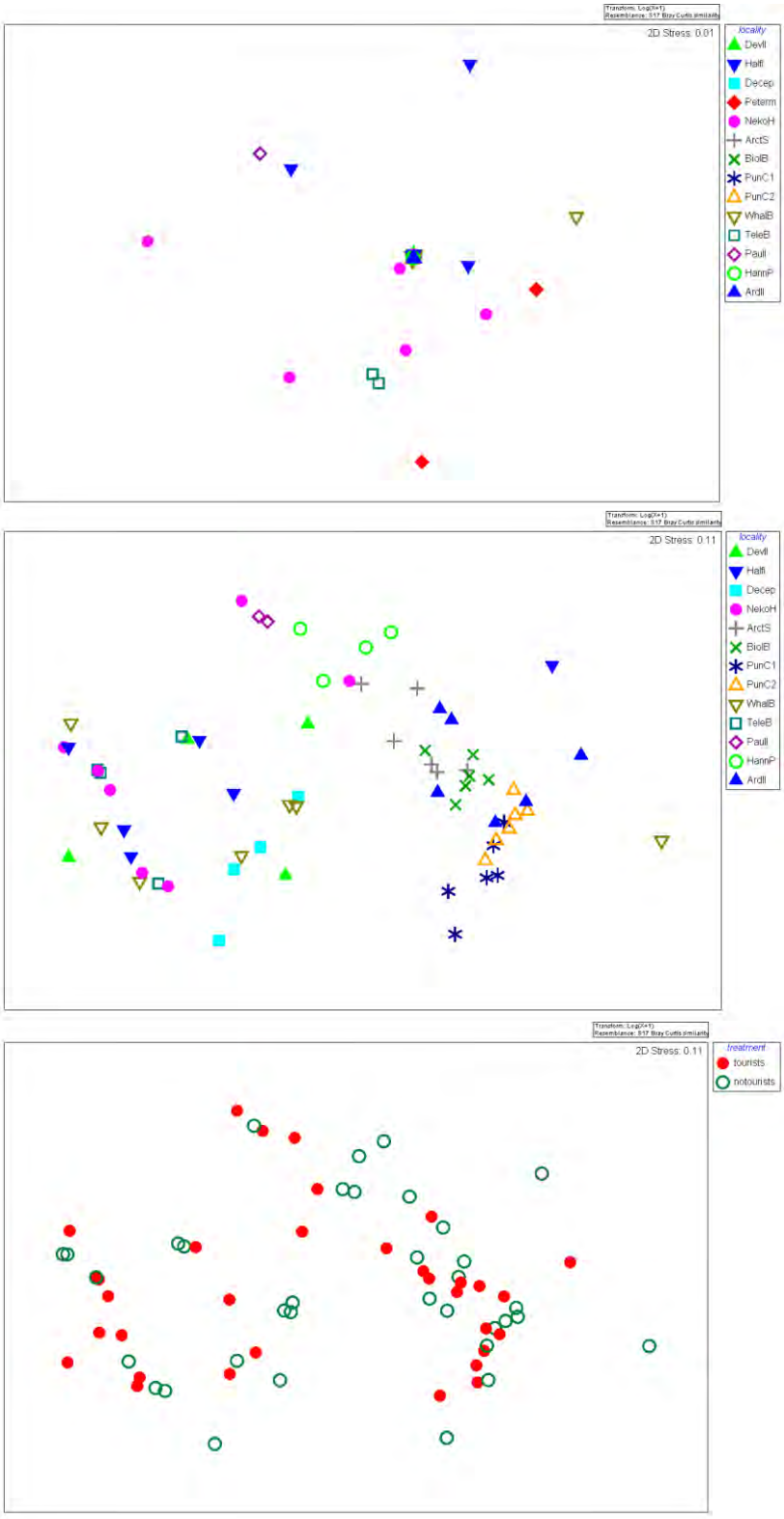


Fig. 176: NMDS-Ordination of the collembolan communities within the studied Antarctic locations. Above: all areas of all localities (labels = "locality"); middle: peripheral areas (= study plots) excluded (labels = "locality"); below: peripheral areas excluded (labels = "treatment" (= human influence)). DevlI: Devil Island, HalfI: Halfmoon Island, Decep: Deception Island, Peterm: Petermann Island, NekoH: Neko Harbour, ArctS: Arctowski Station, BiolB: Biologenbucht, PunC1, 2: Punta Cristian 1 and 2, WhalB: Whalers Bay, TeleB: Telefon Bay, PaulI: Paulet Island, HannP: Hannah Point, ArdlI: Ardley Island.

Table 10: Table of results of a PERMANOVA of the collembolan communities of the studied Antarctic localities. The data can be read as in an ANOVA-Table. df: degrees of freedom; SS: sum of squares; MS: mean squares; F: test statistic; P: significance. Significant results in red.

Source of variability	df	SS	MS	F	P
Locality	12	$1.4*10^5$	12037.0	7.38	0.001
Treatment	1	1243.4	1243.4	0.83	0.540
Locality x Treatment	12	17785.0	1482.1	0.91	0.672
Residuals	42	68513.0	1631.3		
Total	67	$2.3*10^5$			

The test of multivariate homogeneity (PERMDISP) was highly significant for the factor "locality" ($F = 6.997$, $P = 0.001$), but not for the factor "treatment" ($F = 5.307*10^{-2}$, $P = 0.827$). This was to be expected, since areas of some localities ordinated far apart from one another in Fig. 176 (middle), e.g. Whalers Bay, while others grouped very close to each other, e.g. Biologenbucht. The result means that the P value for "locality" cannot be readily accepted, while that for "treatment" can. However, the PERMANOVA result for locality is, in total, nonetheless plausible, since it agrees very well with the NMDS results in that differences between localities were not only marginal, but highly significant.

The plot of a permutative discrimination analysis (CAP) of the collembolan data shows no differentiation concerning the factor "treatment"; anthropogenic influenced and non-influenced plots widely overlap in the diagram (Fig. 177). Accordingly, the cross validation succeeded very poorly: only 57.35% of the areas could be correctly assigned to their treatment level (influenced or not influenced). The canonical test of the influence of the factor treatment was not significant (trace = 0.0476, P = 0.531).

Fig. 177: CAP-Ordination (*canonical analysis of principal coordinates*) of the collembolan communities in the studied Antarctic localities based on the factor "treatment" (= human influence).

3.6.2 Actinedida

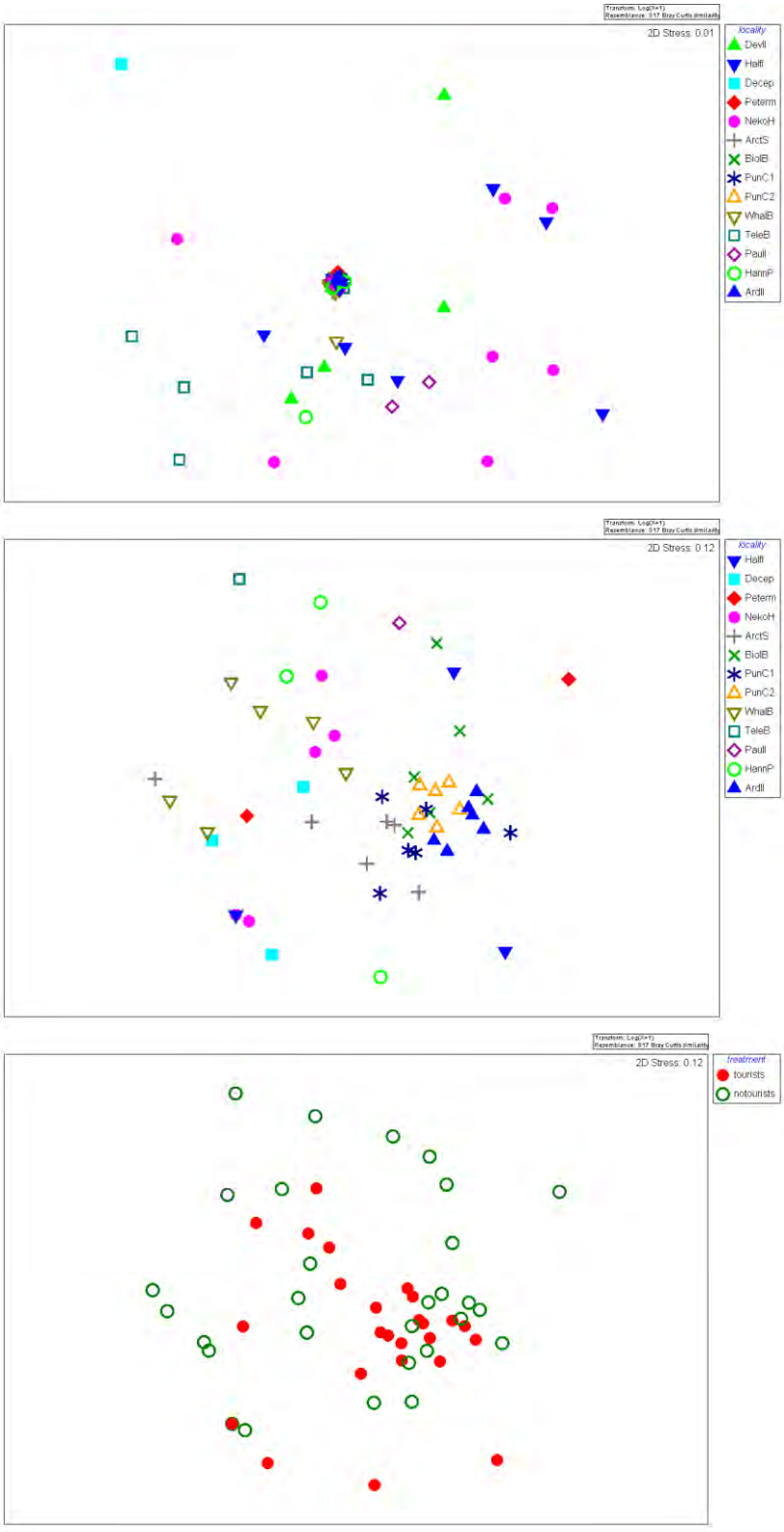


Fig. 178: NMDS-Ordination (*non metric multidimensional scaling*) of Actinedida communities in the studied Antarctic localities. For the arrangement and abbreviations in this figure, see Fig. 176.

As with the Collembola, a NMDS of all Actinedida data resulted in a very tight cluster containing the majority of the study areas, which was surrounded by areas in which only few or no animals were found (Fig. 178, top). If only the areas within the cluster were ordinated, then again a clear influence of locality became visible, but none for human influence (Fig. 178, middle and bottom). These results were confirmed by the PERMANOVA: a highly significant influence of "locality" but none of the factor "treatment". In contrast to the Collembola, the interaction between locality and treatment was highly significant (Table 11). This means that there was indeed a treatment effect, however only in certain localities.

Table 11: Table of the PERMANOVA results for the Actinedida communities of the studied Antarctic localities. For abbreviations see Table 10. Significant results in red.

Source of Variability	df	SS	MS	F	P
Locality	12	89266.0	7438.8	3.33	0.001
Treatment	1	5446.8	5446.8	1.49	0.188
Locality x Treatment	10	39112.0	3911.2	1.75	0.001
Residuals	31	69139.0	2230.3		
Total	54	$2.0*10^5$			

The PERMDISP test of multivariate homogeneity resulted in a high significance for locality (F = 9.344, P = 0.003) and simple significance for treatment (F = 7.305, P = 0.02); data homogeneity is therefore not present within both factors.

The CAP showed for the Actinedida a certain degree of differentiation concerning the factor "treatment"; the "centre of focus" of the distribution of the point cloud lay somewhat dispersed, and about 10 of the anthropogenically non-influenced areas plotted beyond the value range of the anthropogenically influenced point cloud (Fig. 179). The cross validation was more successful than with the Collembola: 64% of the areas could be correctly assigned to their treatment level, for the influenced areas even 77%. The canonical test the influence of the factor treatment was, however, again not significant (trace = 0.2727, P = 0.221).

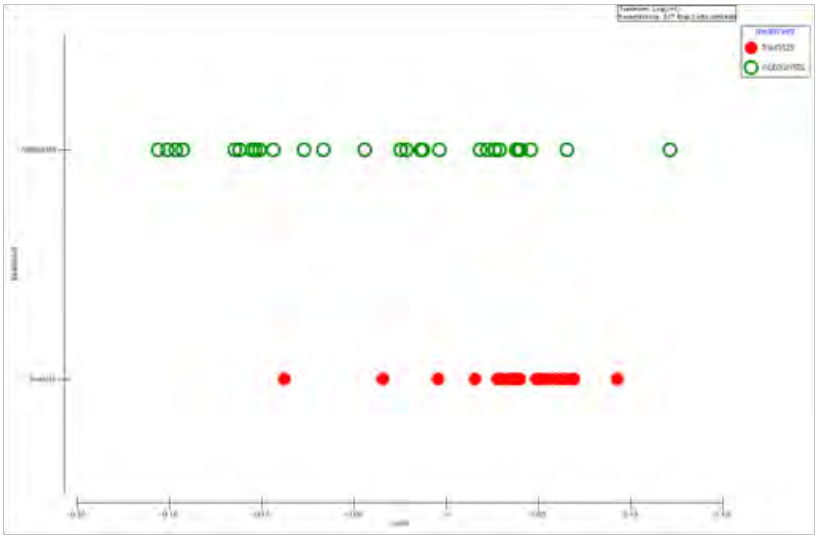


Fig. 179: CAP-Ordination (*canonical analysis of principal coordinates*) of the Actinedida communities of the studied Antarctic localities in regard to the factor "treatment" (human influence).

3.6.3 Oribatida and Gamasida

The data of the Oribatida and Gamasina were analyzed together, since both groups were represented only by few species and in only a few of the localities; a separate analysis would have resulted in non-interpretable patterns.

The NMDS plots were very similar to the animal groups discussed above: a tight cluster of localities when analyzing all dated together; a strong influence of the factor "locality" when "zoomed in" on this cluster; no noticeable influence of human activity (Fig. 5). Somewhat specific for these groups is only that many areas - due to the very low abundances of these groups found there - lay in the periphery of the NMDS plot and the central cluster contained correspondingly fewer areas (Fig. 180, top and middle). This pattern is reflected in the PERMANOVA, which showed a highly significant influence of "locality", but none for "treatment" or the interaction of the two (Table 12). The level of significance for both factors is reliable, since none of the PERMDISP tests were significant (locality: F = 2.932, P = 0.168; treatment: F = 0.864, P = 0.459). Also the CAP could not identify any influence of treatment: the point cloud of anthropogenically influenced and non-influenced areas overlapped widely (Fig. 181); only 51% of the areas could be correctly assigned to the treatment level and the trace statistic was not significant (trace = 0.039, P = 0.698).

Table 12: Table of the PERMANOVA results for the Oribatida and Gamasina communities of the studied Antarctic localities. For abbreviations see Table 10. Significant results in red.

Source of variability	df	SS	MS	F	P
Locality	7	66435.0	9490.7	6.533	0.001
Treatment	1	235.0	235.0	0.240	0.866
Locality x Treatment	6	5767.6	961.3	0.662	0.896
Residuals	24	34865	1452.7		
Total	38	$1.1*10^5$			

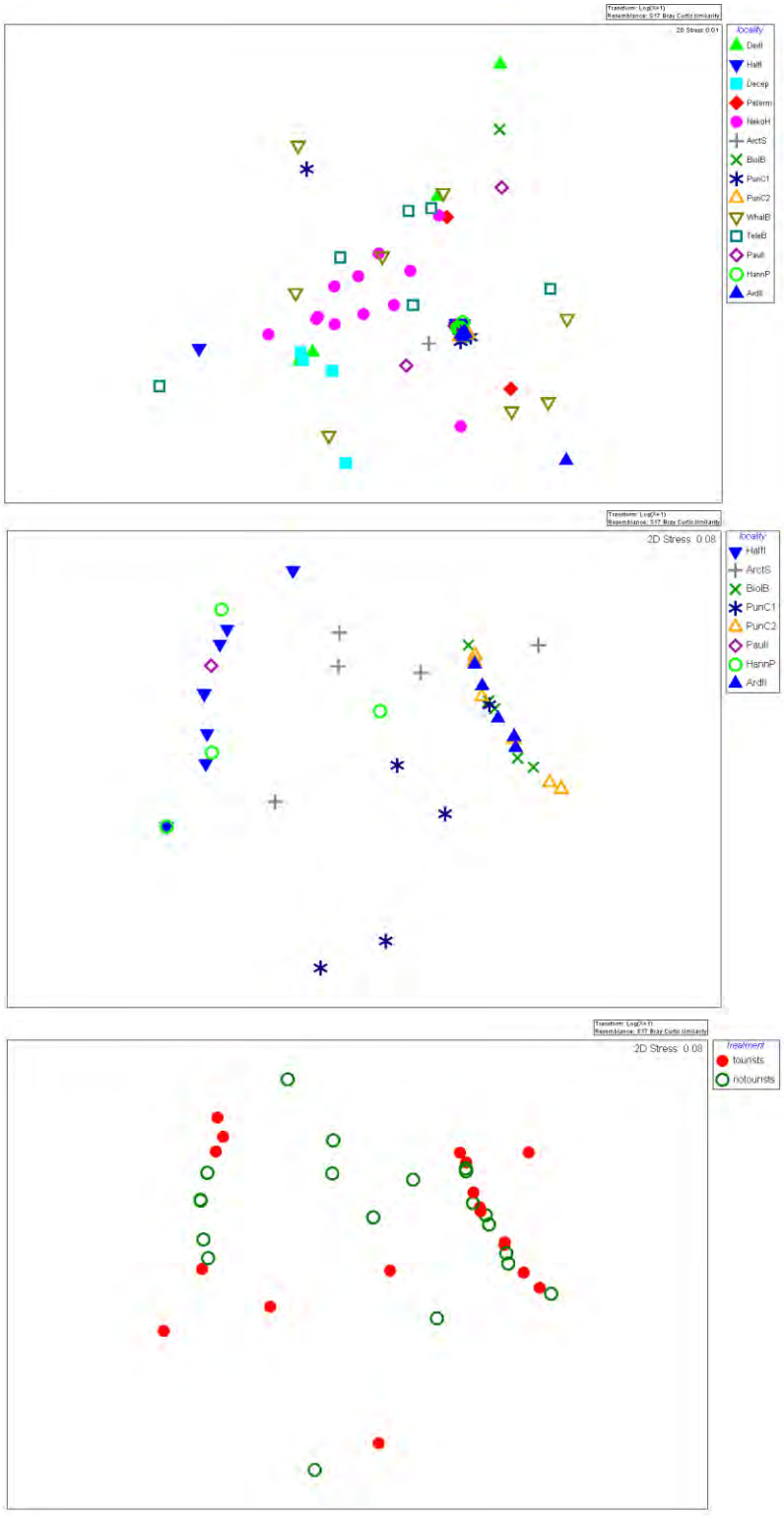


Fig. 180: NMDS-Ordination (*non metric multidimensional scaling*) of the Oribatida and Gamasina communities in the studied Antarctic locations. For arrangement and abbreviations see Fig. 176.

Fig. 181: CAP-Ordination (*canonical analysis of principal coordinates*) of the Oribatida and Gamasina communities of the studied Antarctic localities in regard to the factor "treatment" (human influence).

3.6.4 Nematoda

The community-level analysis of the Nematoda again resulted in (1) a cluster of areas containing large animal abundances, surrounded by areas with much lower abundances (Fig. 182, top); (2) a clear influence of locality, but none of treatment in the ordination of the areas within the cluster (Fig. 182, middle and bottom); (3) corresponding significances within the PERMANOVA (Table 13); (4) a highly significant deviation from multivariate homogeneity for the factor "locality" ($F = 8.086$, $P = 0.001$), but not for "treatment" ($F = 4.406*10^{-3}$, $P = 0.941$); (5) no clear differentiation of the treatment levels in the CAP analysis (Fig. 183, only 8% of areas could be correctly assigned; trace = 0.179, $P = 0.745$).

Table 13: Table of the PERMANOVA results for the nematode communities of the studied Antarctic localities. For abbreviations see Table 10. Significant results in red.

Source of variability	df	SS	MS	F	P
Locality	13	$1.7*10^5$	13125.0	8.126	0.001
Treatment	1	1459.7	1459.7	0.904	0.543
Locality x Treatment	10	16142.0	16142.2	0.999	0.482
Residuals	42	67844.0	1615.3		
Total	66	$2.6*10^5$			

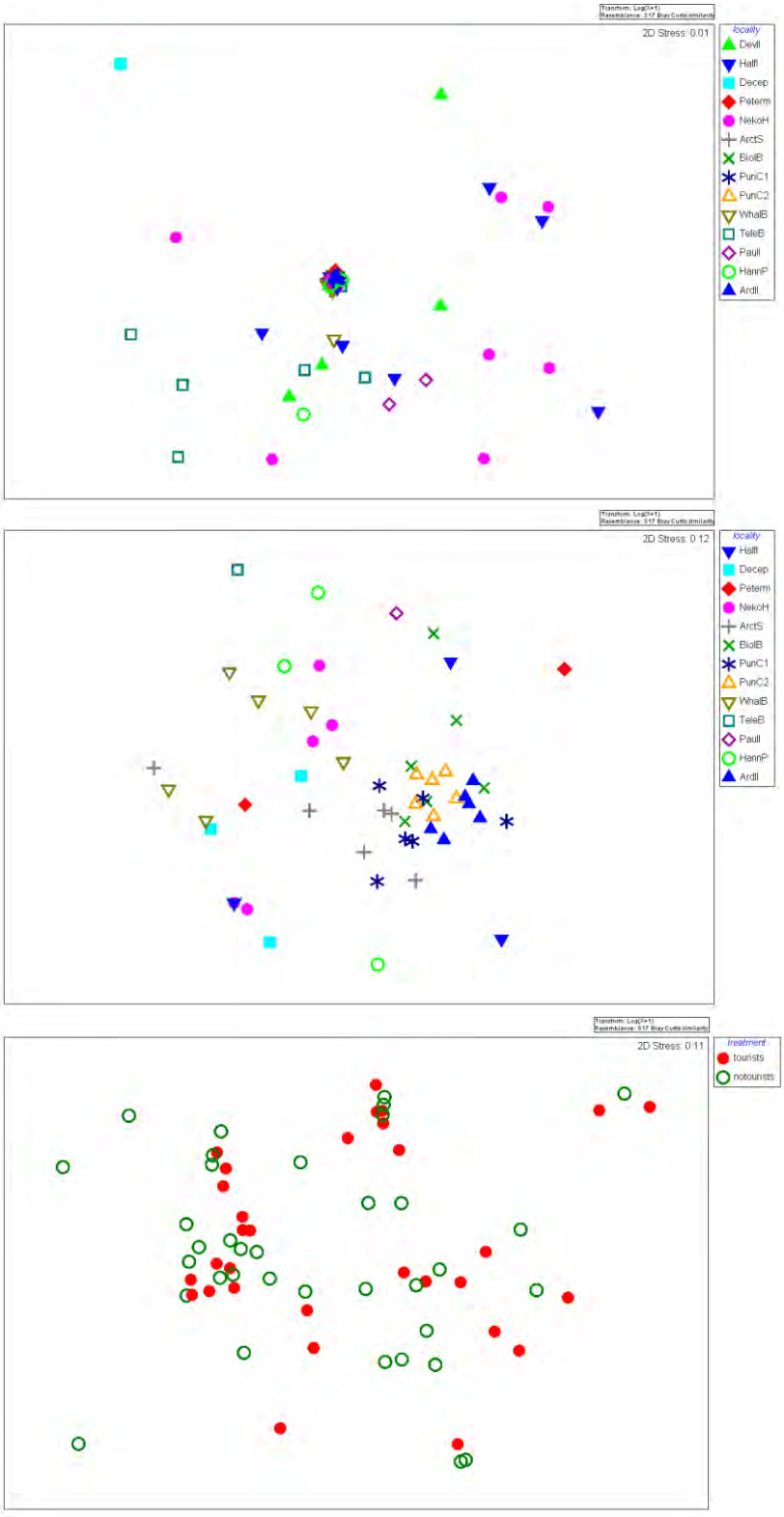


Fig. 182: NMDS-Ordination (*non metric multidimensional scaling*) of the nematode communities of the studied Antarctic localities. For arrangement and abbreviations see Fig. 176.

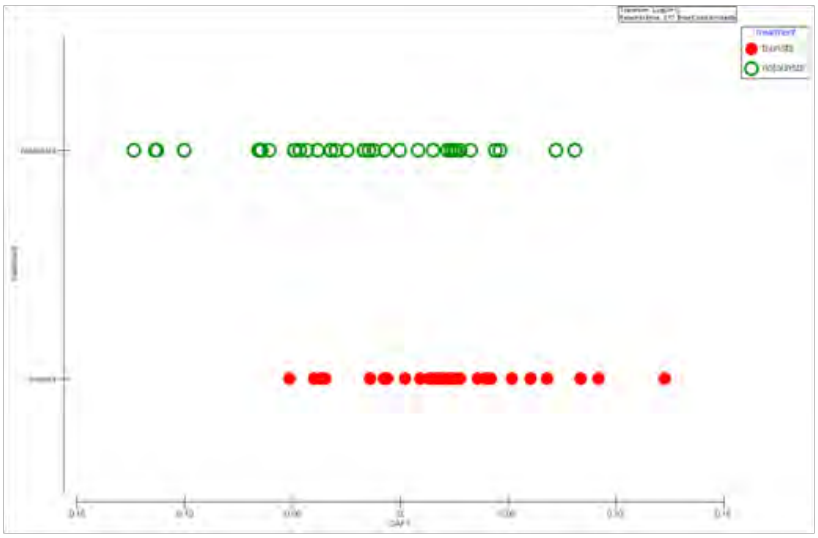


Fig. 183: CAP-Ordination (*canonical analysis of principal coordinates*) of the nematode communities of the studied Antarctic localities in regard to the factor "treatment" (human influence).

3.6.5 Tardigrada

The results of the Tardigrada analyses generally correspond to those of the previously presented animal groups (Fig. 184, Table 14). These analyses also revealed a highly significant deviation from multivariate homogeneity for the factor "locality" (F = 5.061, P = 0.004), but not for "treatment" (PERMDISP F = $1.793*10^{-2}$, P = 0.913). However, it is remarkable that the interaction between locality and treatment was indeed significant (Table 14), which otherwise was only the case with the Actinedida. This indicates that a treatment effect was present, but was limited to only a few localities. The CAP could again not differentiate between anthropogenically influenced and non-influenced areas (Fig. 185); the percent of correctly assigned areas was perhaps as high as with the Actinedida (61.3%), but the trace statistic is again not significant (trace = 0.169, P = 0.110).

Table 14: Table of the PERMANOVA results for the Tardigrada communities in the studied Antarctic localities. For abbreviations see Table 10. Significant results in red.

Source of variability	df	SS	MS	F	P
Locality	13	$1.3*10^5$	10036.0	8.956	0.001
Treatment	1	2576.8	2576.8	0.895	0.448
Locality x Treatment	12	38835.0	3236.3	2.888	0.001
Residuals	35	39221.0	1120.6		
Total	61	$2.2*10^5$			

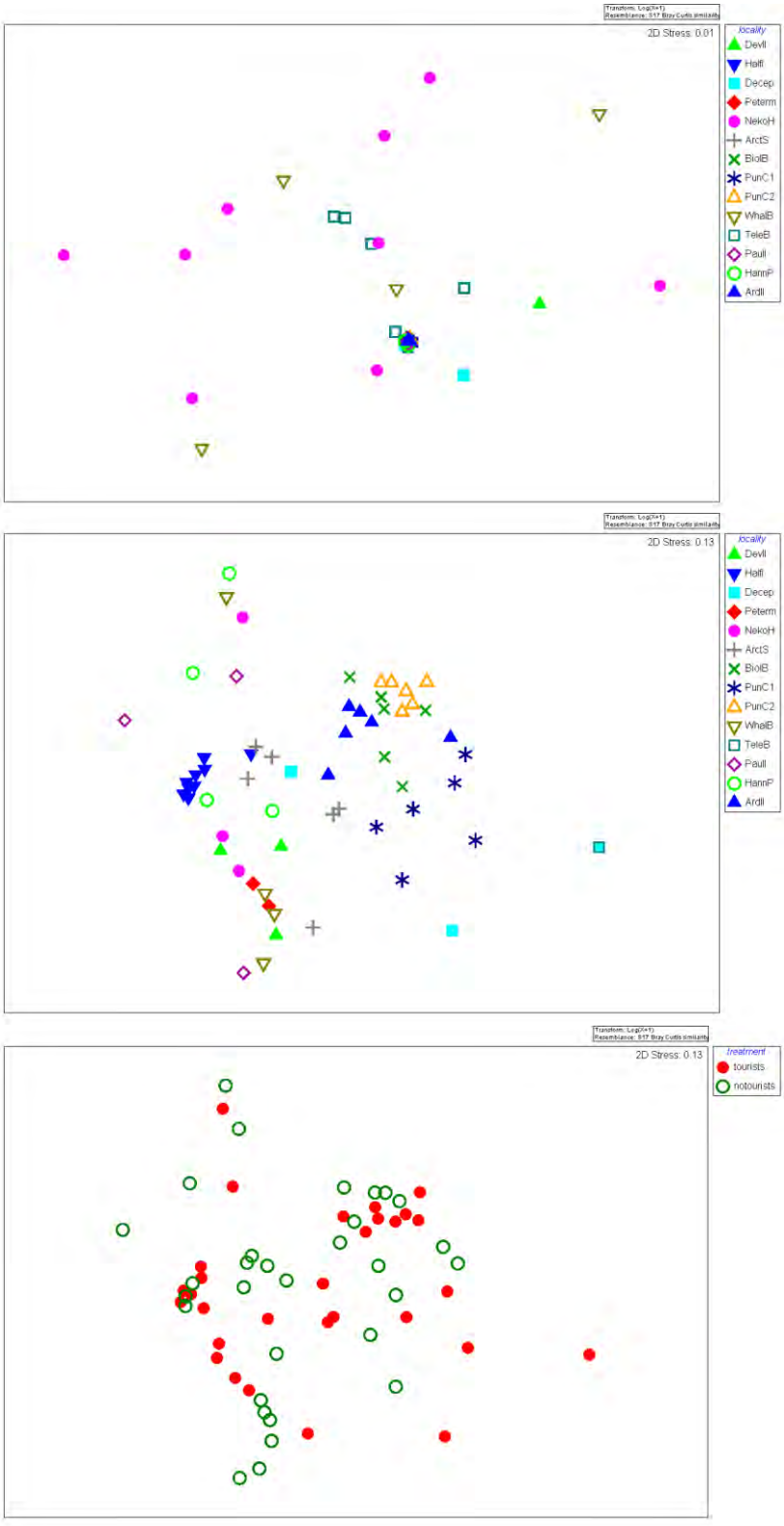


Fig. 184: NMDS-Ordination (*non metric multidimensional scaling*) of the Tardigrada communities of the studied Antarctic localities. For arrangement of the figures and abbreviations see Fig. 176.

Fig. 185: CAP-Ordination (*canonical analysis of principal coordinates*) of the Tardigrada communities of the studied Antarctic localities in regard to the factor "treatment" (anthropogenic influence).

3.6.6 Total fauna (all groups together)

When the data of all studied animal groups were aggregated, then all of the areas can be displayed in a single NMDS plot (Fig. 186, top), since in almost every areal at least a small number of animals were recorded (hardly any zero values). It could be clearly shown for the individual animal groups that the areas of a locality ordinated closely together; the animal communities within a locality are therefore more similar to each other than to those of other localities. The areas did not cluster relative to human influence; the factor "treatment" thus had obviously no effect on the similarity of the soil animal communities (Fig. 186, bottom). This is also shown in the PERMANOVA results: the factor "locality" had a highly significant, the factor "treatment" no influence. As with Actinedida and Tardigrada, the interaction "locality x treatment" was also significant: the effect of treatment (= human impact) depended on the locality (Table 15). However, the level of significance of the PERMANOVA results for the factor "locality" must be interpreted with caution, since the corresponding PERMDISP was significant (F = 16.108, P = 0.001), whereby that for "treatment" was not (F = 0.264, P = 0.655).

Table 15: Table of PERMANOVA results for the combined data of all investigated animal groups from the studied Antarctic localities. For abbreviations see Table 10. Significant results in red.

Source of variability	df	SS	MS	F	P
Locality	13	$1.8*10^5$	14260.0	7.174	0.001
Treatment	1	2100.0	2100.0	0.884	0.569
Locality x Treatment	13	31915.0	2455.0	1.235	0.035
Residuals	54	$1.1*10^5$	1987.7		
Total	81	$3.3*10^5$			

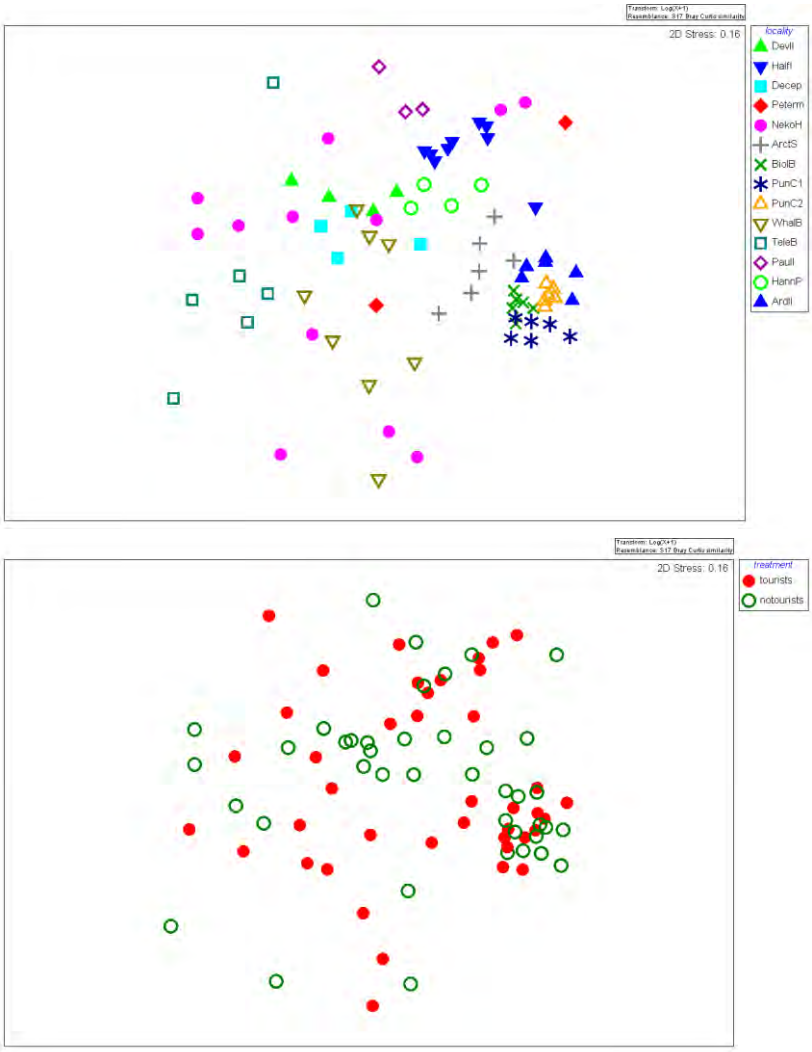


Fig. 186: NMDS-Ordination (*non metric multidimensional scaling*) of the combined data of all investigated animal communities from the studied Antarctic localities. Top: areas labelled according to locality; bottom: areas labelled according to human influence. For abbreviations see Fig. 176.

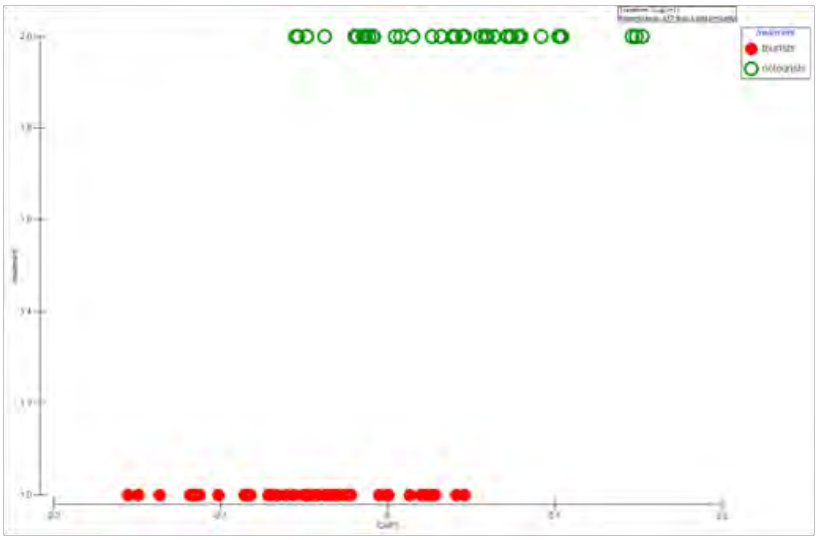


Fig. 187: CAP-Ordination (*canonical analysis of principal coordinates*) of the combined data of all investigated animal communities from the studied Antarctic localities in regard to the factor treatment (human influence).

3.6.7 Summary of the multivariate analyses

The multivariate analysis of the present Antarctic data set was usually difficult, since it contained many zero values originating from animal-free soil samples. Since almost all of the procedures used here are sensitive against zero values, it was necessary to resort to certain statistical "tricks": firstly, not the single soil-sample data were analyzed, but rather the samples of the single areas were combined for the calculations (whereby information of the lowest hierarchy level of the sampling design became lost). Secondly, since in some animal groups individuals were not recorded from all areas, the analyses focused on "positive findings". Even if this procedure is statistically correct, it results in a limitation of the available information. Although the analysis results often suggested that "something was there", the ordinarily very effective multivariate techniques used here proved to be unusually inefficient.

Among all the studied animal groups, even with aggregated data sets, the factor "locality" had an outstandingly large and significant influence on the similarity of the communities. The closer the study areas were to each other, the more similar was the species composition and dominant structure of the animal communities. The factor "treatment" (human influence yes/no), on the other hand, did not play a significant role according to these analyses. The only permissible scientific conclusion is therefore that historical or recent human activities were not very relevant for the soil animal communities of the studied sites (at least concerning the aspect community similarity).

However, not really fitting into this conclusion are the Actinedida, Tardigrada and the combined data (= all studied animal groups combined). Although with these groups also no significant human influence could be discerned with the multivariate analyses, the interaction "locality x treatment" was indeed significant. For the Actinedida and combined data (and also for the Tardigrada, but less clearly), the CAP analysis further showed a shift in the treatment cluster (*nota bene*: but also not significant!). It can therefore be presumed that at an impact of human activity on community diversity was present in at least some areas of some of the localities. Due to the sparse data, however, this cannot be analyzed in detail with community statistical procedures. The questions "where does an impact exist?" and „ how does it affect the

communities?" must remain unanswered until further, more adequate sample material is available.

3.6.8 Impact on the β-Diversity of the soil animals

To test whether humans had a significant impact on the β-Diversity of animal communities, a PERMDISP on the presence/absence-transformed data of all groups was performed. The test could not be used for five localities, since less than three areas were sampled in these localities. For none of the nine tested localities could a significant human impact be proven (Table 16).

Table 16: Results of the test of significant differences in β-Diversity between anthropogenicaly influenced and non-influenced areas. Some localities could not be tested, since less than 2-3 areas were sampled there. t: test statistic; P: significance of the resuls.

Locality	T		P
Arctowski Station	0.195		1.000
Biologenbucht	1.627		0.214
Deception Island		No test possible	
Devil Island		No test possible	
Halfmoon Island	0.094		0.994
Neko Harbour	0.626		0.612
Petermann Island		No test possible	
Punta Cristian 1	1.857		0.402
Punta Cristian 2	1.929		0.212
Ardley Island	1.561		0.190
Hannah Point		No test possible	
Paulet Island		No test possible	
Telefon	2.101		0.496
Whalers Bay	0.950		0.593

3.7 Assessment of the efficiency of the bootwashing procedure on board the MS Hanseatic

Large differences existed between the two sets of "control cleansing" samples. The wash water was strongly contaminated after the land excursion on South Georgia. The profiles of the boots contained soil and penguin feces etc. (mostly, but not only) between the tread grooves of the soles, which was washed out with the control wash water. Despite conservation with (in the end) 50% ethanol, a strong growth and activity of microorganisms (mostly bacteria, which however could not be quantified) could be documented. In 30% of the samples, remains of feathers and detritus could be found, while soil particles (mostly sand grains) were present in 60% of the samples. Plant remains (presumably moss leaves etc.) were found in more than 50% of the samples. Soil animals were also discovered in somewhat more than 50% of the samples (Table 17). Due to the strong bacterial activity, these were usually fairly decomposed, so that their identification to species level was not possible. In one case, however, a fully intact nematode individual could be proven with certainty.

In contrast, the samples after the land excursion on Deception Island were relatively clean. Although soil particles (again mostly grains of sand) were registered in almost all cases (80% of the samples); however, microbial activity was only exceptionally (10%) detected. Remains of feathers and plants were present in 20% of the samples. Two nematode individuals were recorded in two separate samples. Their identification was also somewhat uncertain due to their strong decomposition.

Table 17: Total number of individuals found in the samples of the „control cleaning" of passengers' footwear after passing through the "bootwashing" equipment. In no sample was more than one individual detected.

Sampling date	Excursion goal	Actinedida	Oribatida	Nematoda	Tardigrada
1 Februray, 2011	Salisbury Plain (South Georgia)	1	1	1	2
7 February, 2011	Deception Island	0	0	2	0

4 Discussion

4.1 General faunistical characteristics

4.1.1 Microarthopods (= Mesofauna)

Of the various Antarctic soil invertebrate groups, microarthropods (Collembola, Acari) have previously been most intensively studied. This background allows an in-depth comparison between earlier studies and the present results, which is therefore given here especially for these animal groups. Based on this comparison, the microarthropod fauna (= mesofauna) found in the present investigations can be considered typical for the maritime Antarctic. Almost all of the recorded species have been previously found in Maritime Antarctica. A number of the registered taxa are also endemic to Antarctica, e.g., the collembolans *Cryptopygus antarcticus*, *Friesea grisea*, *Folsomotoma octooculata* and the acarines *Globoppia loxolineata*, *Nanorchestes berryi*, *N. gressitti*, the *Eupodes parvus* subspecies or the *Apotriophytydeus* species. Close to 90% of the known continental Antarctic species and almost 50% of the maritime Antarctic mesofauna are endemic (Marshall & Pugh 1996, Hogg & Stevens 2002). The level of endemism, however, is lower at the generic level. Whereby a major proportion of the genera occurring in the continental Antarctic are endemic, most of the genera found in the maritime Antarctic are cosmopolitan and even resemble the Arctic fauna (Marshal & Pugh 1996, Strandtmann 1967). However, except for introduced species, even the non-endemic species found in the present study are mostly limited to the southern oceans (Pugh 1993). Examples of more widely dispersed species can be found among the Oribatida (Acari). All of the oribatids recorded in this study exhibit a broad distribution in the Antarctic and Subantarctic, which can reach to New Zealand (*Alaskozetes antarcticus*) or South America (*Liochthonius mollis*). Nonetheless, as far as can be ascertained from literature reviews (Block & Starý 1996, Starý & Block 1998, Pugh 1993) and other sources (e.g., Convey & Smith 1997), especially the oribatid species *Alaskozetes antarcticus*, *Globoppia loxolineata* and *Halozetes belgicae* belong to the typical species inventory of the region studied here.

The endemic species and even the entire species composition of soil-animal communities occurring in Continental and Maritime Antarctica are generally different (Hogg & Stevens 2002). The faunas of the two areas are highly separated by a biogeographical boundary between Continental Antarctica and the Antarctic Peninsula (the so-called Gressitt Line: Convey et al. 2000b, Chown & Convey 2007, Convey 2011). At such large spatial scales, a main factor influencing species' distribution are historical events (colonization/ recolonization of refugia), whereby the dispersal capability of the various species appears to be an important factor of colonization. The origins of the extant species in the two regions are apparently different: species occurring in Continental Antarctica appear to be Gondwanan relicts of populations present before glaciation of the continent, while species of the Antarctic Peninsula are probably often post-glaciation (= post-Pleistocene) colonists (Marshall & Pugh 1996, Convey & Smith 1997, Convey 2001, Convey & Stevens 2007, Caruso et al. 2009). The present investigations thus conclusively recorded no microarthropod species typical for the continental Antarctic, further reinforcing the conclusion that the registered fauna is highly characteristic for the maritime Antarctic.

In terrestrial habitats of the Antarctic, very large site-to-site differences in species composition are common (e.g., Wise et al. 1964, Tilbrook 1967b, Richard et al. 1994, Sohlenius et al. 1995),

which was also found in the present study. However, very few species are restricted to specific sites (= local endemics). Also the present study found no local endemics. Furthermore, no additions to the zoogeographical distribution of the recorded species could be found. Previous studies show that the distributional areas of almost all species extend further north (e.g., to the South Orkney Islands) as well as much further south within the maritime Antarctic (e.g., Palmer Land, Graham Coast or even Alexander Island; see the Results sections of the individual species above). Nonetheless, the soil fauna of many of the investigated localities had never been surveyed before. Therefore, the number of proven sites of distribution for most of the recorded species has been increased by the present study, especially within the South Shetland Islands.

Although a number of different locations were examined during these investigations, the study areas were somewhat limited in range within the maritime Antarctic. Nonetheless all Collembolan species endemic to Maritime Antarctica were recorded in the present study. Only very few non-native species known to occur were not found here. The "basic species set" (as defined here, see Results section of Collembola) of the maritime Antarctic (*Cryptopygus antarcticus, Friesea grisea, Folsomotoma octooculata*, *Cryptopygus badasa, Tullbergia mixta, Friesea woyciechowskii*) were registered on almost all locations on and around King George Island (this set is formally incomplete only in Punta Cristian, where *C. antarcticus* is lacking). In contrast, not all mite species known to occur on and around the Antarctic Peninsula were registered in the present study, most likely due to the restricted study area, e.g., the oribatid *Magellozetes antarcticus* or the actinedid *Rhagida leechi*. Moreover, some widely distributed taxa often found abundant in previous studies were only found sporadically in the present investigation, e.g., the oribatids *Globoppia loxolineata* and *Liochthonius mollis* or the actinedids *Pretriophtydeus tilbrooki*, *Rhagidia gerlachei* or *Stereotydeus villosus*. Furthermore, although the collembolan species were registered in relative proportions similar to other studies, the abundances and dominances of closely related mite taxa were recorded in opposite proportions to those found in previous investigations, e.g. the actinedids *Eupodes parvus* and *Eupodes exiguous* or *Nanorchestes berryi* and *N. nivalis*. This is partly due to the fact that densities of individual species vary strongly from sample to sample and their distribution is very patchy even at short distances (Richard et al. 1994, Ohyama & Shimada 1998). The most plausible explanation for these different dominances is, however, that the substrates sampled in the present studies were different than those generally sampled during general species inventories. Many basic surveys of Antarctic microarthropods simultaneously investigate different microhabitats (e.g., algal mats, vegetation and stones) or even largely concentrate on the underside of stones (e.g., Janetschek 1967, Goddard 1979a, Convey et al. 1996, Thor 1996, Convey & Quintana 1997, Stevens & Hogg 2002). Many authors consider the underside of medium-sized stones to be an important habitat for terrestrial microarthropods in Antarctica. This microhabitat is characterized by, e.g., a greater availability of soil moisture or organic carbon and tend to heat strongly in summer and retain temperatures often more than 10°C above air temperatures, allowing the animals to be more active while at the same time avoiding exposed microhabitats (Wise et al. 1964, Tilbrook 1967b, Caruso & Bargagli 2007, Hawes et al. 2008). Many species aggregate and are therefore found in higher densities under stones, whereas other, mostly smaller species are more commonly found in soil substrates (Wise et al. 1964, Bowra et al. 1966, Goddard 1979a, Booth & Usher 1986, Caruso & Bargagli 2007). However, sampling stones can only be achieved by hand by experienced researchers and is time intensive. Most of the sampling in the present study took place during land excursions from cruise ships, which are highly limited in time. The sampling design employed here also

required equivalent samples from both human influenced and non-influenced plots, not allowing specific microhabitats to be directly addressed in single samples. Furthermore, sampling often took place by inexperienced personnel, requiring a straightforward and standardized sampling design. Thus, although the soil cores were not necessarily always the best habitats for Antarctic microarthropods (especially for studies of ß-diversity), the investigation of soil substrates (including surface vegetation) was necessitated by the research questions.

Table 18: Microarthropod species numbers recorded in previous studies in maritime Antarctic localities.

	Study	Region/Locality	Microarthropods	Collembola	Acari	Actinedida	Oribatida	Gamasida	Astigmata/Other
Maritime Antarctica	Russell et al. 2012 (present study)	N. maritime Antarctica	**45**	11	34	25	5	4	
	Tilbrook 1967	maritime Antarctica	**22**	7	15	7	5	2	1
	Gressitt 1967	N. maritime Antarctica	**37**	5	32	5	4	2	0
	Wallwork 1973	maritime Antarctica	**33**	8	25	10	15	0	0
	Block 1984	maritime Antarctica	**40**	8	32	10	14	4	4
	Pugh 1993*	maritime Antarctica			70	27	20	9	14
	Marshall & Pugh 1996	maritime Antarctica			17				
	Convey 2001	maritime Antarctica	**46**	10	36				
	Hogg & Stevens 2002	maritime Antarctica	**47**	15	32	13	15	4	0
Region	Convey et al. 2000	South Sandwich Islands	**19**	8	11	2	6	3	0
	Gryziak 2009	South Shetland Islands			28	13	12	2	1
	Usher & Edwards 1986 cit. in Convey & Quintana 1997	South Shetland Islands	**17**						
Individual sites	Strong 1967	Palmer Station	**14**	4	10	4	5	1	0
	Goddard 1979	Signy Island			10	6	2	1	1
	Goddard 1979b	Signy Island			18	8	2	4	4
	Block 1982 cit. in Richard et al. 1994	Signy Island	**13**	3	10				
	Usher & Booth 1984	Signy Island	**10**	4	6	5	0	1	0
	Usher & Edwards 1984	Lynch Island	**13**	3	10	7	2	1	0
	Usher & Edwards 1986 cit. in Convey & Quintana 1997	Marquerite Bay	**11**						
	Richard et al. 1994 & Convey et al. 1996	Byers Peninsula Livingston Island	**21**	6	15	9	5	1	0
	Convey & Quintana 1997	Cierva Point, Danco Coast	**15**	3	12	6	4	1	1
	Convey & Smith 1997	Marguerite Bay	**20**	4	16	9	6	1	0
	Convey & Smith 1997	Alexander Island	**9**	2	7	6	1	0	0
	Convey et al. 2000	Charlot Island	**7**	0	7	3	4	0	0

Antarctic soil faunal communities are generally known to be species poor and lacking major taxonomical groups common in more temperate climates (e.g., Carabidae, Lumbricidae, Diplopoda etc.; Block 1984b, Marshall & Pugh 1996, Hogg & Stevens 2002). Over the last decades, the number of microarthopod species known from the maritime Antarctic has increased steadily (Table 18). Recent publications list up to 47 microarthropod species

occurring naturally in the maritime Antarctic (e.g., Convey 2001, Hogg & Stevens 2002). In an intensive literature review, Pugh (1993) even lists approx. 70 different terrestrial species of the Acari having been found in the maritime Antarctic, although these also include synanthropic taxa occurring around research stations etc. With over 40 registered species (plus 4-8 potentially non-native species, see below), the present study thus recorded a large proportion of the maritime Antarctic microarthropod fauna.

The mesofaunal species richnesses found in the individual localities generally correspond to the species numbers recorded in previous studies in other sites (cf. Table 18). The number of species found in maritime Antarctic habitats is generally much lower than those of temperate zones (cf. Pertersen & Luxton 1982), and are often even only half of that found in Subantarctic sites (Pugh 1993, Hogg & Stevens 2002). On the other hand, the species richnesses found the present study were larger than those recorded in most continental Antarctic study sites, which can often be an order of magnitude lower than maritime Antarctic habitats (Convey et al. 2000b).

The reasons for such low species numbers are considered to be - besides the geographic isolation of Antarctica - the necessity of occurring species to be adapted for survival under the adverse climatic Antarctic conditions (Gressit 1961, Block 1984b, Marshall & Pugh 1996). Various adaptations are found in Antarctic species, the most obvious of which being an ability to withstand very cold temperatures. Most species actually show a high range of temperature tolerances (Sanyal 2004), but are nonetheless very cold tolerant, being able to withstand temperatures down to -20°C and even lower (e.g., Tilbrook 1967b, Janetschek 1967, Block 1979, Day et al. 2009). This is usually achieved by super-cooling abilities, such as avoidance of ice nucleators (including emptying the gut) and/or antifreeze proteins in the body (Block 1984b, Lister et al. 1988, Hogg & Stevens 2002, etc.). Some species are actually incapable of tolerating warmer temperatures, i.e., above 20°C (Janetschek 1967).

A further adaptation is an extended life cycle relative to species from temperate climates (Goddard 1979a, Block 1984b, Booth & Usher 1986, Lister et al. 1988, Marshall & Pugh 1996). Generation times can be as long as 2 to 3 years (as opposed to ca. 1 year in temperate zones), whereby species can overwinter in various stages and eggs can hatch either immediately or in following years (Janetschek 1967, Strong 1967, Goddard 1979a, Booth & Usher 1986). This has been considered to be an adaptation to unpredictable yearly weather conditions (Booth & Usher 1986) and can lead to highly variable densities from year-to-year. Many of these adaptations can also be found to various degrees in related species from temperate climate zones and thus are likely to be pre-adaptations in taxa ultimately colonizing Antarctica (Marshall & Pugh 1996, Hogg & Stevens 2002). The necessity for such adaptations limits the numbers and identities of the species occurring in Antarctic terrestrial habitats.

Some of the localities sampled in the present study were - in comparison to other studied localities or to previous studies in the maritime Antarctic - found to be extremely species poor (e.g., Paulet Island, Devil Island or Petermann Island). Since the same sampling design was used on these islands as in all other localities, this species poverty can be considered to be a general characteristic of these islands. The islands were extremely rocky with very thin and/or poorly developed soil substrates. Vegetation was also hardly or not at all developed on these islands; however, other localities also bore no vegetation and nonetheless showed a higher species richness. Thus, the lack of soil substrate (with the corresponding organic material and microorganisms) was the most likely cause of the low species richness in these localities.

Among the microarthropod groups, Actinedida were the most species rich, followed by Collembola. This also appears to be typical for terrestrial maritime Antarctic arthropod faunas. Most previous surveys of soil fauna in the maritime Antarctic report Actinedida (usually called Prostigmata in the literature) with the most species, followed by Collembola and then the other mite groups (see citations in Table 18). This is in contrast to temperate regions, where microarthropod communities are mostly dominated by Collembola and Oribatida.

Only few previous studies report total average densities of Acari, Collembola or microarthropods (Table 19). A comparison of these studies show a large range of registered densities, generally averaging between 2,000 and 50,000 individuals m^{-2} with maximum densities at times reaching more than 400,000 individuals m^{-2}. A comparison of these literature values with the present study is difficult, due to the necessary standardization of all samples to individuals $100cm^{-3}$ in the present analyses. Nonetheless, considering an individual sample size of 5 cm Ø and 5 cm depth (which was the most common sample size, with approx. half of each sample used for extraction of microarthropods), the density values calculated here can be very roughly translated into individuals m^{-2} (Table 19). Although these values can only be considered rough approximations, they show that the densities found in the present studies generally compare to those found in previous studies in the maritime Antarctic. As opposed to species richness, the recorded densities were comparable and often higher than those generally found in temperate climates (cf. Petersen & Luxton 1982). In many localities, the densities found here can actually be considered to be very high, often due to aggregations of individual species. The extremely high densities found in, e.g., the second locality at Punta Christian (King George Island), Halfmoon Island or Whalers Bay on Deception Island (all in 2010) were indeed due to high densities of single species. Convey & Smith (1997) consider heat extractions for sampling microarthropods as used here to provide low individual numbers, thus underestimating actual densities or giving contradictory results. This cannot be confirmed in the present study. The islands proving to be very species poor mentioned above (or also Telephone Bay on Deception Island) also generally showed low densities. However, this was true in all samples of these localities, so this tends to be a general characteristic of these islands, most likely for the same reasons mentioned above. As opposed to species richness, the highest densities were mostly found among the Collembola, which also appears to be common in Antarctic microarthropod communities (Table 19).

The results acquired in the present project clearly showed a decline in diversity (densities and species richness) at higher latitudes, although this was not as distinct when literature data was included, e.g., in the collembolan analyses (s. Results). Whether literature data was included or not, the fauna of Deception Island is obviously the species richest, which appears to be an anomaly, but can be explained by the special anthropogenic and climatic conditions as well as its history (see discussion of non-native species below). Declines in diversity at higher latitudes also appears to be typical for the maritime Antarctic soil fauna, as a number of authors have also remarked on this tendency (e.g., Usher & Edwards 1986b, Convey & Quintana 1997, Caruso & Bargagli 2007, Gryziak 2009). A clearer decline of collembolan diversity was shown at a larger scale by Usher and Edwards (1986b), where the number of species decreased from 5 to 3 on islands of the maritime Antarctic from the north-east to south-west (South Shetland Islands - Graham coast - northern Adelaide Island). That this cannot be regarded as a general rule, however, has been remarked upon, since surveys in more southern regions of the maritime Antarctic have revealed relatively species-rich microarthropod communities (much species

richer than continental sites at the same latitude; Convey & Smith 1997). Thus, this tendency may be only true for the northern maritime Antarctic, in which the present investigations took place. As an exception, this tendency was not found within the Oribatida. However, relative to many previous studies, the Oribatida were actually only sporadically registered in the present investigations. This is most likely due to the differently sampled substrates as mentioned above, whereby oribatid species are very often found in high abundances on or under stones or in sites with a stronger vegetation cover (Goddard 1979a, Block 1984b, Convey & Smith 1997), which were not preferentially sampled within the present study.

Table 19: Microarthropod densities recorded in previous studies in maritime Antarctic localities (as far as available). Numbers in individuals per m^2. The densities given for the present study are generalized transformations from individuals $100cm^{-3}$ and are only rough approximations. True densities are given in Appendix 3.

Study	Locality	Microarthropods	Collembola	Acari
Tilbrook 1967	maritime Antarctic	2,000-45,000 (max: 78,000)		
Goddard 1979	Signy Island			1,300-28,000
Block 1982 cit. Richard et al. 1994	Signy Island	20,000-99,000		
Usher & Edwards 1984	Lynch Island	11,000-29,000 (max: 68,000)		
Usher & Booth 1984	Signy Island		8,000-50,000 (max: 107,000)	4,000-30,000 (max: 96,000)
Richard et al. 1994 & Convey et al. 1996	Byers Peninsula (Livingston Isl.)	<1,000-21,000 (max: 46,000)	(dominant)	
Convey & Smith 1997	Alexander Island	240-3,000 (max: 20,500)		
Convey & Smith 1997	Marguerite Bay	43,000-121,000 (max: 433,000)	(dominant)	
Convey & Quintana 1997	Cierva Point, Danco Coast	5,000-50,000 (max: 83,000)		
Convey et al. 2000	Charlot Island			12,000-44,000
Russell et al. 2012	Arctowski Station	236,000	116,000	120,000
(Present study)	Biologenbucht	255,000	206,000	50,000
	Punta Cristian	69,000	25,000	44,000
	Punta Cristian II	288,000	247,000	41,000
	Ardley Island	101,000	64,000	36,500
	Halfmoon Island (2010)	730,000	15,000	717,000
	Halfmoon Island (2011)	49,000	670	48,000
	Hannah Point	20,000	1,900	18,000
	Telefon Bay	1,900	1,100	900
	Whalers Bay (2010)	347,000	346,000	1,600
	Whalers Bay (2011)	145,000	124,000	21,000
	Paulet Island	9,900	300	9,600
	Devil Island	6,600	6,600	0
	Neko Harbour (2010)	140	50	90
	Neko Harbour (2011)	5,700	3,600	2,100
	Petermann Island	275	0	275

The presence results also indicate a high year-to-year variability both in densities and species richness. A number of authors have also reported a high variability of microarthropod quantities within the same site at longer time scales (years), which appears to be dependent

upon year-to-year differences in precipitation and temperature (e.g., Goddard 1979a, Usher & Booth 1984, Ayres et al. 2007). Thus, the temporal variability observed in the present study is also not unusual for the Antarctic terrestrial fauna.

At the larger scale of the maritime Antarctic, the total densities of mesofaunal groups as well as densities of particular species are determined mainly by locality. This is confirmed in the present study, where the most significant differences in densities and species richness of all taxonomic groups was between localities. Such a dependence is similar to the large-scale distribution pattern described in eastern Antarctica, where the presence of a species is strongly affected by colonization/recolonization of refugia periodically covered with ice (Caruso et al. 2009, Stevens & Hogg 2003, 2006). If true, local species richness varies considerably depending on the geographical position of the locality. The large-scale distribution pattern in the maritime Antarctic is probably more influenced by climatic determinants than their history, since almost all indigenous species are distributed widely throughout the Antarctic Peninsula. In the present study, the highest densities of Collembola and Acari were found on King George Island and Deception Island. However, this is not confirmed by other publications, where high densities were also found in more southern latitudes (e.g., Tilbrook 1967a, Usher & Edwards 1986b). We suggest that such strong differences between locations are caused by local differences at medium scales (characteristics of the particular coast, exposition of slope, soil parameters, development of vegetation, etc.).

In total, all of the parameters mentioned above characterizing the registered microarthropod communities show very typical relationships for Antarctic soil faunas. Therefore, the data obtained in the present sampling can be considered very representative for the studied habitats. This exemplifies that the database used for determining an anthropogenic influence on the soil fauna is sound.

Before discussing human influence on the soil follows, it is necessary to mention the observed dependence of taxonomical groups and single species on various habitat parameters, since a human influence can affect these parameters, which in turn can cause changes in the soil fauna (indirect effects). Due to the wide distribution of most species occurring in the maritime Antarctic, some authors consider Antarctic species to have a low habitat specificity and correspondingly broad tolerance for a wide range of habitat conditions, indicating a more generalist nature (Tilbrook 1967b, Richard et al. 1994, Convey & Quintana 1997). The correlation analyses performed here cannot substantiate this opinion. Most major taxonomic groups (total densities and species richness) and many individual species showed significant relationships to specific habitat parameters.

At the local scale of patchy environments within the Antarctic, abiotic factors (e.g., soil properties and nutrient status) appear to be more important than biotic ones in influencing microarthropod assemblages (Adams et al. 2006, Hogg et al. 2006). The dependencies found in the present study mostly confirmed the known regularities of the microhabitat distribution of Antarctic mircroarthopods. In general, soil moisture is often a limiting factor affecting the distribution and abundance of species (Wise et al. 1964, Dalenius 1965, Strandtmann et al. 1967, Strong 1967, Ohyama 1978, Block 1984b, Booth & Usher 1984, Frati et al. 1997, Hogg & Stevens 2002, Sinclair et al. 2006, Day et al. 2009). More than directly affecting species distributions, temperature can more often determine the amount of biologically available water. Nonetheless an upper limit of soil moisture (about 12%) exists for most species. Different species can also show different tolerances for water-logging or dry conditions (Hayward et al.

2004). The present results confirm a potential limitation by soil moisture, since all major taxonomic groups and many individual species showed significantly higher densities (and species richness with the major groups) at higher soil moistures.

The strongest correlations among microarthropods were to vegetational cover. In Antarctic habitats, a strong dependence of total and individual species' densities on vegetation cover has been shown by, e.g., Tilbrook (1967b), Goddard (1979a), Usher & Booth (1984), Booth & Usher (1986), Richard et al. (1994), Convey et al. (1996), Frati et al. (1997) and Gryziak (2009). This could likely be an indirect effect, since few of the recorded species are directly herbivorous. Hogg & Stevens (2002) consider vegetation to be more a source of habitat than of food for Antarctic Collembola and Acari. On the other hand, vegetated areas are often richer in microorganisms (bacteria, fungi, algae) due to the photosynthetic activities of plants. Since almost all of the recorded species are microbivorous (cf. Strandtmann et al. 1967, Goddard 1979b, Goddard 1979a, 1979b, Block 1979), vegetation could likely be a habitat with richer nutritional resources (cf. Sinclair (2001). Independent of the mechanisms involved, however, the study results show the degree of vegetation cover to be a very important habitat factor determining densities and speeches richness.

For most groups and species, soil organic matter (total amounts, concentrations of C and N and or C/N-relationship) was an important determinant factor in the present study. The dependency of soil Collembola and Acari on soil organic matter has been confirmed in a number of studies (e.g., Wise et al. 1964, Booth & Usher 1984, Sanyal 2004, Adams et al. 2006). Although this is true for most of the groups studied here, the correlations were strongest among the Oribatida in the present study. This is not surprising, since most Oribatida are considered to be - as opposed to almost all other soil microarthropods - particulate detritus feeders (Krantz & Walter 2003), which is probably also true of the oribatid species occurring in Antarctica. Gryziak (2009) found Oribatida to be limited to older, more developed soils, which are generally more enriched in organic matter. Therefore, on the one hand, vegetated soil substrates with a higher content of dead organic matter most likely offer a spatially and climatically more hospitable habitat than predominantly mineral sand, gravel or rock substrates. On the other hand, soil organic matter represents a primary nutrient resource for Antarctic Oribatida. Again, organic matter may be an indirect, but nonetheless important influencing factor, providing a more substantial basis for microorganisms, which in turn provide nutrient resources for the soil fauna.

The weakest correlations to habitat parameters were found with the wholly predacious Gamasina. Except for vegetational cover, abiotic soil parameters were apparently not as important in determining the occurrence of this mite taxon as in the other faunal groups. Prey availability is most likely the more important determining factor (Lister et al. 1988), although this was not specifically studied. Other soil factors, such as pH or soil texture (= grain size distribution), showed few or no correlations to densities or species richness of Gamasina. This may be due to the fact that such correlations truly do not exist. However, it is more probable that the limited number of study sites and the usually very low densities of this animal group (as party also true among the Oribatida) render the statistical determination of actually occurring relationships to habitat parameters difficult. On the other hand, a low influence of these factors on soil microarthropods has also been found in other studies (e.g., Wise et al. 1964, Adams et al. 2006). Therefore, the most important habitat factors determining species

richness and population sizes of the mesofauna in the present study were vegetational cover, soil organic material and soil moisture.

4.1.2 Microfauna

Due to the relative paucity of studies on the nematode and tardigrade fauna of the maritime Antarctic, a comparison of the present results with literature data is far more difficult than for the mesofauna. Nonetheless, such a comparison confirms many phenomena described above for the mesofauna (species richness and densities; dependence on locality, geography, habitat parameters, etc.) and are not repeated here in detail.

Despite fewer previous studies than for the mesofauna, compared to other, even European areas, the Antarctic nematode fauna has been fairly well studied during the last 40 years (e.g., Spaull 1973, Maslen 1979a, 1979b, Andrassy 1998, Holovachov & Boström 2006). However, a some previous Antarctic studies did not investigate soil-borne nematode communities, but concerned the microfauna from moss cushions and lichens. The 14 studied localities and over 300 soil samples in the present study thus represent a relatively extensive investigation and provide a considerable data source.

In a biogeographical context, Antarctica is unique in regards to its nematode fauna for two reasons. Firstly, most of the nematode species having been recorded from Antarctica are endemic or at least unknown elsewhere. Secondly, hardly any overlap in nematode species inventory exists between Maritime and Continental Antarctica; these two regions also represent separate biogeographical zones for Nematoda (Maslen & Convey 2006). An intersection of the two zones could be Alexander Island, which is species richer than neighbouring islands to the north and even harbours 10% more species than the rest of Maritime Antarctica together (Maslen & Convey 2006). First evidence of the occurrence of supposedly continental Antarctic nematode species in the maritime Antarctic was given by Maslen & Convey (2006) from Adelaide Island, Alamode Island und Charcot Island, where nematode specimens were found that morphologically strongly resembled the continental species *Plectus murrayi* and *P. frigophilus*. Also the present study produced records of a nematode species from various maritime Antarctic islands, which is morphologically very similar to the continental Antarctic species *Panagrolaimus magnivulvatus*, and has accordingly been given here the nominal appellation *Panagrolaimus* cf. *magnivulvatus*. It remains to be clarified whether these records truly represent the continental species *P. magnivulvatus*, which would thus indicate a species' distribution that transgresses the maritime Antarctic - continental Antarctic border (Gressitt line), or whether these specimens simply represent a new species. In the later case, the distribution of *Panagrolaimus* species would resemble that of the genus *Eudorylaimus* (Andrássy 2008), with a clear distributional separation between maritime Antarctic and continental Antarctic species of the genus. In turn, it cannot be clarified here how a possible transport across the border of these two biogeographical zones could take place. In light of the very minimal touristic or scientific traffic between Maritime and the Continental Antarctica, transfer via clothing or equipment of tourists or scientific personnel (see below) appears unlikely.

The number of total species that have been previously reported in Antarctica (= gamma diversity) is very small in comparison to other climatic zones of the planet (Wharton 2003). Andrássy (2008) only lists 32 known species in the maritime Antarctic. The present study already recorded 22 to 23 nematodes species alone in the individual richest sites

(Biologenbucht, Arctowski Station and Punta Christian I on King George Island). In the species-poorest locality (Peterman Island), on the other hand, only one species was found. The species composition of the nematodes fauna is thus very different from island to island. This high regional ß-diversity suggests barriers to species' distributions, which could result from deficient nutrient resources or ecological conditions or simply be due to physical barriers that cannot be crossed (Spaull 1973).

Spaull (1973) and Maslen (1979a, 1979b) studied the nematode fauna in soil and vegetation samples from 15 and 16 maritime Antarctic islands, respectively, among which only Deception Island was also studied in the current project. These authors determined that nematode diversity - measured either as the number of genera or species per site - decreased with increasing southern latitude. The authors supposed a relationship between diversity and decreasing temperatures from north to south, with the corresponding increase in abiotic stress as well as the decreasing ice- and snow-free periods, in which colonization and population establishment of nematode species can take place. The present study could confirm the decrease in species diversity from north(east) to south(west), albeit in a much smaller region of the maritime Antarctic. Nematode diversity was furthermore associated with the degree of vegetation cover, which also at least in trend decreased from northeast to southwest. On the other hand, nematode diversity was also associated with soil water and nutrient contents, which differed strongly among the various localities, but did not show a gradient from north to south. While the present study could thus confirm the tendency of decreasing species richness with increasing southern latitude, the causal relationship behind this tendency is more likely due to site-specific environmental conditions, which differentially affected the occurrence of the nematode species.

Terrestrial Antarctic nematodes feed primarily on bacteria, cyanobacteria and algae. The nematode species recorded in the present study were clearly dominated by bacteriovores, which in trend increased in the climatically harsher south-western study sites in the Weddle See. Bacterivores also represented the most numerous feeding type among the nematodes of the climatically milder localities on King George Island and Ardley Island. However, in these sites, this feeding type was also accompanied by algivorous and fungivor-radicivorous species. Microbial biomass (e.g., as a nutrient resource for nematodes) is generally low in Antarctica due to low soil nutrient contents (Andrássy 1998). An exception is represented by the ornithogenic soils on Paulet Island, from which an outstandingly strong population of one bacterivorous nematode species (*Rhomborhabditis* cf. *teres*) was detected in the present study. These very high densities of one species are most likely due to the (relative to most Antarctic soils) high nitrogen content and presumed correspondingly rich source of bacteria. Sohlenius et al. (2004) also observed that representatives of the bacterivorous genera *Plectus* and *Panagrolaimus* developed their highest population densities in organic soils and mosses as well as in the organic material under the algae *Prasiola*. In turn, such microhabitats were particularly found in the vicinity of colonies of the snow petrel (*Pagodroma nivalis*), whose droppings provided a rich nutrient input into the soils.

Due to the usually sporadic or lacking vegetation cover and the correspondingly weak soil food webs, only few plant-parasitic and predatory nematode species are found in Antarctic soils (Andrássy 1998). Exceptions are represented by as yet mostly undescribed species from the genera *Tylenchus* (from Signy Island), *Filenchus* (from Livingston Island, King George Island and Ardley Island) as well as *Aphelenchoides vaughani, A. helicosoma* and *A. haguei* (all

occurring from Signy Island to Alexander Island), all of which feed on plant-root and fungal cells (Spaull 1973, Maslen, 1979a, Chipev et al. 1996 and the present study). Also the predatory nematode species *Coomansus gerlachei* is broadly distributed in Maritime Antarctica (Peneva et al. 1996), with at times astonishingly high densities despite the minimal supply of prey, as could be shown in the present study on Halfmoon Island and Hannah Point. The high dominance of representatives of this species in the nematode communities of these two localities indicates that they do not feed primarily from other nematodes species, but rather (or also) from Rotatoria, Protozoa and juvenile Tardigrada.

The soil habitat studied in the present project is a relatively unexplored environment for Tardigrada. There are a few published research papers reporting tardigrades in soils, which are very limited for the Antarctic and nothing on the scale of the current project. This study has explored (for tardigrades) new sites and habitats around the maritime Antarctic and therefore has increased the knowledge of species richness for some of the localities. King George Island had been reported with 15 species of tardigrades (Dastych 1984, Jennings 1979), to which this study adds: *Hexapodibius boothi*, *Isohypsibius* sp., *Diphascon* (*Diphascon*) sp. and *Calohypsibius* cf. *ornatus*. Most of the study sites at King George Island were new and the soil habitat previously unexplored, which will have contributed to this data. Three species had been previously reported for Deception Island – *Acutuncus antarcticus*, *Ramajendas renaudi* (≡ *figidus*) and *Dactylobiotus* (cf.) *ambiguus* (Downie et al. 2000), to which *Hexapodibius boothi* and *Macrobiotus* cf. *furciger* have been added by the present study. The specific sampling sites for these results do not overlap with earlier studies, which included two from permanent aquatic pools and the third from *Prasiola crispa* on a boulder. Other study sites do not have any direct comparison with published data and the current results therefore represent new knowledge of tardigrade distribution in the Antarctic.

The limited overlap of study sites over the two years also made it difficult to draw comparative conclusions. Halfmoon Island showed the greatest similarity between years for species richness and population numbers, but with more areas providing tardigrades in the samples in 2011. The results from Whalers Bay on Deception Island distinctly varied between the two seasons, with very low numbers recovered in 2011. Similarly, Neko Harbour provided tardigrades in the samples from the anthropogenically influenced sites in 2010, but nothing from these sites in 2011.

An element apparent in all the samples, but not shown in the statistical analyses was the difference in life-cycle stages. Samples from the 2011 collection showed substantially larger ratios of juveniles to adults. This could be related to the time period of the collections, differences in severity of the previous winter and/ or the time of spring thaw. Also evident in the samples were the fewer numbers present in 2011 over 2010. Again, this could reflect the weather patterns for the period, but the limited overlap of study sites made this difficult to quantify. Curiously, *Ramajendas* cf. *frigidus* was found hosting large numbers of peritrichs (= commensal sessile ciliate protozoa) in the 2010 samples, but there were fewer and/or damaged peritrichs in the 2011 samples. It is possible the extraction technique was more robust in 2011, causing some of the peritrichs to be damaged, but most appeared to be absent from the host (no remaining holdfasts). It is possible that seasonal variations changed the life-cycle of one or both host and commensal.

4.2 Anthropogenic influence

4.2.1 Influence on the existing native soil fauna

One of the most important hypotheses behind the present study concerned whether human influence has a direct impact on terrestrial invertebrate populations and thus on the individual numbers and species richness of the Antarctic soil fauna. In the present study, anthropogenic influence was studied at the species level and in many locations not previously studied. Since no data on soil organisms in the studied sites was available before human activities began, a temporal before/after comparison in the same plots was not possible and only spatial comparisons between anthropogenically influenced and non-influenced areas could be carried out. The results attained in these investigations could definitively show affects of human presence (= in general, trampling affects since no human waste or other contamination is allowed or known to occur in the study areas). However, the results were not persistent throughout all groups, study years or faunistic parameters. Thus, anthropogenic effects on the existing soil fauna were at times subtle, "cryptic" within the total data and often masked by high data variability (both within localities and especially between localities). Exact statistical procedures were therefore necessary to discern true human impacts. Nonetheless, the statistical analyses could show that a significant anthropogenic influence on the studied soil fauna does indeed exist.

At a higher taxonomic level, anthropogenic pressure significantly affected primarily the total densities of the various taxonomic groups, with the exception of Actinedida. In many animal groups significant effects were sometimes found in only one of the study years, but these groups usually showed an overall effect when the data of both study years was analyzed together. Only the Gamasina were an exception to this rule, probably due to the low total individual numbers registered in this mite group. Collembola and Nematoda showed significant effects in both study years (Nematoda in the year 2010, however, when only the localities on and around King George Island were regarded). In all significant cases, the effects were negative, with lower total densities in the anthropogenically influenced areas. Although not persistent throughout all taxonomic groups and often not in all study years, these consistently negative results (when significant) clearly indicate that human trampling pressure does have a damaging effect on the maritime Antarctic soil fauna.

An anthropogenic effect on species richness was generally not found. Although for Collembola and Oribatida significantly reduced species richness in the human influenced areas could be determined in single years (2011 and 2010, respectively), these were more the exception. For all other animal groups, no significant influence in species richness was observed, and no overall effect in species richness for both years could be found in any of the major groups. Therefore, human trampling negatively affected the abundances of the occurring fauna, but not noticeably the number of occurring species.

The most important result of the covariance analyses (ANCOVAs) was detection of the fauna's dependencies on vegetation cover, with both higher total densities and species richness with increasing vegetational cover. That vegetation cover apparently has a positive effect on total densities and species richness was true for almost all taxonomic groups, which was also confirmed by the results of the correlation analyses. Both microfaunal groups, Nematoda and Tardigrada, showed a positive influence of vegetation in both years, while the mesofaunal groups generally exhibited this dependency on vegetation in only one year or the other. The

post-hoc analyses of the influence of vegetation showed increasing densities and species richnesses at all levels of vegetational cover. This indicates the importance of vegetation for Antarctic soil fauna, regardless of how developed the vegetation is. This dependency of Antarctic soil fauna on vegetation has been remarked upon by many authors (e.g., Usher & Booth 1984, Richard et al. 1994, Convey et al. 1996, Frati et al. 1997 and Gryziak 2009). Surprising was especially the lack of or minimal significant relation to vegetational cover by the Oribatida, although they significantly correlated with soil organic matter. Thus, again, organic material in soils is apparently more important in determining the distribution and abundances of Oribatida than the vegetation itself.

Furthermore, the ANCOVAs indicated a strong interaction between human influence and vegetation cover. The most common effect was human influence being stronger and negative in plots with medium levels of vegetation cover. In other words, in many locations, when vegetation was only sporadic, human impact was significantly more negative than if no and/or much vegetation was present. Despite the increasing species richnesses with increasing vegetational cover, as in the variance analyses, species richness was independent of human trampling at all levels of vegetational cover (again, with the exceptions mentioned above). This dependence of human influence on vegetational cover is an important result, since in Antarctic habitats generally only larger stands of closed vegetation are protected in ASPAs or as "no-go" areas in the Visitor Site Guides of specific localities. The experiences of the project team in touristically visited localities showed that areas with only sporadic or moderate vegetational cover (as well as local collections of organic matter or moisture, i.e. in and around melt streams or lakes) were generally ignored by the expedition teams directing the tourist traffic. Since the present results indicate that the soil fauna in these "ignored" areas are actually preferentially negatively influenced by human activity, the anthropogenic damage to the soil fauna can be locally (at the microhabitat scale) even much greater than the general results suggest.

In this study, the main differences in the soil faunas existed between localities, showing that the most important determinants of the faunas are environmental factors, which are probably more influential than a human impact itself. However, human impact depended on local conditions and under certain conditions (e.g., sporadic or moderate vegetational cover, melt streams) could actually be much larger than the overall effects of all study areas together. This indicates the importance of a stronger regulation of human activities based upon the local conditions of each individual location.

Importantly, the effects on individual taxonomic major groups were not additive. Due to the fact that Antarctic "soils" are so species poor (see discussion above), it was suspected that human effects could be more easily discerned when the species of all animal groups were cumulatively analyzed as "total soil fauna". This was not the case. It thus appears that human trampling differentially affects the different animal groups and for different reasons, most likely indirect effects due to a human influence on habitat parameters specific to a locality, which then in turn affect the soil fauna.

Only very few previous studies investigated anthropogenic effects on the Antarctic soil fauna, and almost none of these at the species level. In both experimentally controlled and natural non-controlled situations, Tejedo et al. (2008) studied the influence of human trampling on three factors: resistance to compression, bulk density of soils and total density of Collembola in areas of human activity over five summer field seasons on the Byers Peninsula (Livingston Island, South Shetland Islands). Only vegetation-free soils were studied. These authors

demonstrated that even a minimal human presence is sufficient to alter both physical and biological characteristics of Byers Peninsula soils, although at the lowest levels of human activity this difference was not significant in comparison with adjacent undisturbed control areas.

Some data on human impact were also obtained by Bulavintsev (1990) on the Fildes Peninsula (King George Island, South Shetland Islands). Collembola were scarce in sites close to research stations frequently used by caterpillar transport, on roads and on a platform used for short storage of kitchen garbage. Only single specimens of *C. antarcticus* were found in areas of strong human impact. Areas with less significant disturbance were more occupied by Collembola. *Cryptopygus badasa* (= *Cryptopygus* sp. in Bulavintsev 1990), *Folsomotoma octooculata, Cryptopygus antarcticus* and *Tullbergia mixta* (= *Tullbergia* sp. in Bulavintsev 1990) were found there, but the general abundance of the collembolan assemblage was relatively low (several tens of individuals per core). Furthermore, Ayres et al. (2008) could show in the McMurdo Dry Valleys that a relationship exists between human activities and the number of Antarctic nematode individuals. These authors ascribed these effects to a mechanical compression of the soil substrates and reduction in the amount of pore space representing the habitat of endogeic soil animals. Their study was all the more remarkable, since even very low human influence (only few researchers per year) was found to cause negative impacts, possibly due to the extreme conditions (lack of moisture and organic material, very simple soil biotic communities) in these continental Antarctic sites. The present study thus generally confirms the results of these previous studies. It furthermore shows that anthropogenic impacts do not necessarily only directly affect the animals themselves, as the case with mechanical compression of soil substrates shows. Moreover, a human influence – as described above – also contained indirect, site- and species-specific components and was closely related to the degree of vegetational cover.

The human influence on individual taxa was species specific, with some species being negatively influenced (lower abundances at higher human influence), while a few other species were actually positively influenced (higher abundances in human influenced areas). Such species-specific reactions to various forms of disturbance or environmental changes are actually commonplace among soil faunal communities. Such species-specific effects were found regardless of the statistical analyses used (ANOVA or ANCOVA). The ANCOVA analyses could also show higher abundances with increasing vegetation cover among many individual species. The only exceptions to this were among potentially non-native species (see discussion of these below). These analyses further revealed significant interactions between human influence and vegetational cover for a number of species, with a significantly larger human influence (be it positive or negative) at medium or sometimes high vegetational cover. This could be due to the general trend of higher individual densities with higher vegetational cover, thus allowing stronger populations to be differentially more impacted by human influence. However, this does not explain why some species are actually more influenced at medium levels of vegetational cover (25-75%), although their densities were highest at the strongest levels of vegetational cover. This may be due to somewhat weaker populations being more susceptible to direct human influence or humans indirectly influencing less-developed habitat conditions, among many other possibilities. Most important in this regard, however, is – as mentioned above - that medium levels of vegetational cover are not regarded by expedition leaders or research personnel to warrant protection against human trampling. Only closed vegetation is

avoided. If species within medium levels of vegetation are actually more affected by humans, then this lack of regard can be potentially disastrous on the long-term (human effects in such regularly visited tourist or research locations must be considered to be increasingly cumulative).

It remains very speculative why such species-specific reactions occur. On the one hand, negative reactions can be both direct or indirect, with human trampling causing direct mortality and thus reductions in species' populations or indirectly causing changes in, e.g., soil conditions or nutrient resources, which thus negatively affect a species' population. The evidence suggests the possibility of changing habitat conditions, e.g. changing the soil structure by compaction and thus the moisture content and internal spaces in which these soils organisms live, and thereby favouring or hindering different species. On the other hand, positive influences on total densities or individual species' abundances are most likely indirect, i.e. reduced competition (due to smaller populations of negatively affected species) or changes in habitat conditions, leading to an actual increase in the species' preferred habitat or nutritional requirements. The very species-poor Antarctic soil food webs are very sensitive to habitat changes such as increasing soil moisture, nutrient input or temperatures (Freckman & Virginia 1997, Barrett 2008). Freckman & Virginia 1997 presume this sensitivity to be due to the lacking functional redundancy among the constituent species combined with a particular sensitivity of specific species to environmental changes, which render the communities as a whole very susceptible to human impacts.

Furthermore, for instance among the Tardigrada, soil dwellers or species tolerant of the soil environment appear to be particularly affected by anthropogenic influence. Those species more requiring greater vegetation cover and being less soil-bound are also negatively associated with human impact, but probably via an indirect impact on the vegetation itself. Although not explored by analysis, tourist impact could have an indirect affect on life cycles and post-winter population recovery by altering the habitat, e.g. paths being cleared of snow and ice, thus potentially exposing habitats to increased run-off from surrounding areas, faster drying or soil compaction. There is a suggestion in the results obtained here that some of the anthropogenic impact may be linked to the time of tourist arrival and its relationship to the actual start of the Antarctic summer season.

Considering all animal groups together, the number of species which exhibited either a significant positive or negative reaction to human influence reached > 35% of the species occurring in large enough populations to allow a statistical analysis. If species showing a statistically non-significant, but stronger tendency for impacts by humans are included, almost 2/3 of all analyzed species show an impact in one direction or the other (Fig. 188). Although not specifically analyzed, such major changes of the populations within a species community strongly indicate a considerable alteration within the structure of the soil food webs in those areas experiencing anthropogenic influence. This, in turn, can significantly influence biotic interactions within the communities. Biotic interactions have been considered a strong determinant of species occurrences within the generally species-poor soil communities of the Antarctic (but see Hogg et al. 2006). Since these species-poor communities also lack functional redundancy typical for the communities of, i.e., temperate habitats, a constant human influence of tourism and research personnel could lead to major changes in the ecological functions of the soil biota altogether (cf. Adams et al. 2006, Barrett et al. 2006, 2008, Wall et al. 2006). Such alterations in soil food-web ecological function could be even more important for

Antarctic terrestrial ecosystems than, i.e., significant reductions in total densities or species richness.

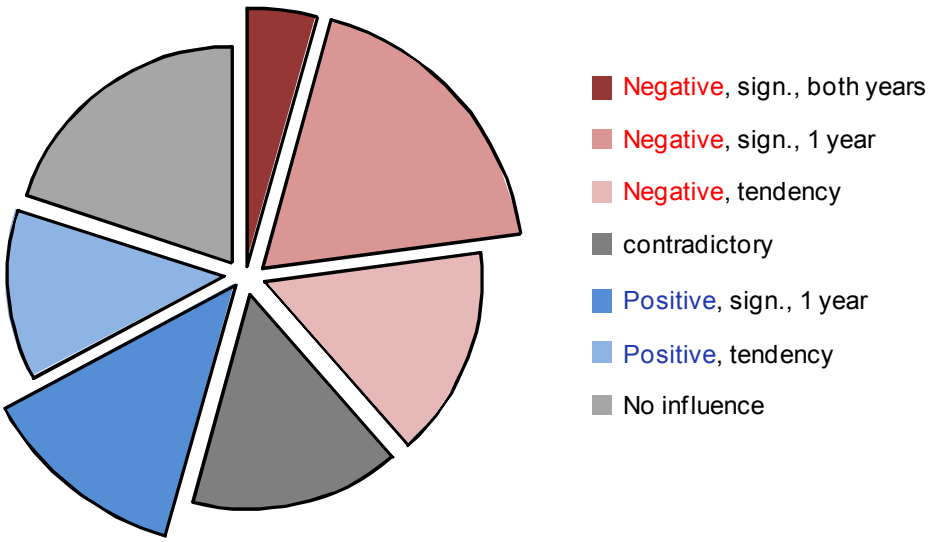


Fig. 188: Proportion of all analysed species showing either negative or positive effects of anthropogenic influence. "sign." = statistically significant effects in either one or both study years (no positive effects were found for any species in both study years). "tendency" = species showing statistically non-significant effects, but a strong tendency for either reduced or increased populations in human influenced areas. "contradictory" = species exhibiting significant effects, being positive in one of the study years and negative in the other.

4.2.2 Influence on the ß-diversity of the Antarctic soil fauna

A second hypothesis of the current study concerned a possible "homogenisation" of the soil-faunal populations among and within a locality due to human activities. One possibility of homogenisation is a transfer of species from locality to locality (reduced regional ß-diversity). In the species' populations studied here, no evidence of this form of population inter-site homogenisation could be found. Most of the recorded species (as an example, among the Tardigrada: *Acutuncus antarcticus*, *Macrobiotus* cf. *furciger*, *Diphascon* cf. *pingue* and *Hypsibius* cf. *dujardini*) are relatively ubiquitous throughout the maritime Antarctic (e.g., Jennings 1976). These species' population dynamics and composition is dependent on, e.g., vegetation and moisture. Other species have been less frequently observed or have been missed by favourably collecting moss and lichens rather than soils and therefore no knowledge is available to determine whether these species are being moved. Species such as *Echiniscus meridionalis* and *Hexapodibius boothi* are still local to the original type localities, which would suggest that inter-island transportation is limited or negligible. Similar can be said for the species of the other animal groups. Especially the microarthropod species are widely distributed throughout the maritime Antarctic, so that no changes in the geographic distribution were observable in the present study. Detection of an anthropogenically caused spread of species among different localities is further compounded by the fact that the distributions of many species is highly variable even between sites of similar habitat characteristics (e.g., Gressitt 1967, Tilbrook 1967b, Ohyama & Hiruta 1995), which was also found in the present study.

In local habitats with many different microhabitats (such as found in the Antarctic), the species composition of various microhabitats within a locality can be different. This plot-to-plot change

in species composition is a general source of biodiversity (= local ß-diversity). Very high spatial heterogeneity within the Antarctic soil fauna has been reported in a number of studies, often with well-developed communities found close to areas with no soil fauna at all (Ohyama 1978, Richard et al. 1994, Stevens & Hogg 2002, Adams et al. 2006, Caruso & Bargagli 2007, Simmons et al. 2009). High local species turnover and the discontinuous distribution of Antarctic soil fauna is often due to local patches containing no soil fauna at all, this being often between 25 and 50% of samples (Wise et al. 1964, Sohlenius et al. 1995, Adams et al. 2006). A further source of ß-diversity in Antarctic soil fauna is often very high densities due to aggregations in single points within a locality (Usher & Booth 1984, Sohlenius et al. 1995, Convey & Quintana 1997, Convey & Smith 1997).

It is possible that tourists or station personnel transport soil substrate on their boots or clothing and that soil animals are thereby transferred from microhabitat to microhabitat. This could lead to an increase in the local distribution of the species, resulting in changes in the diversity of microhabitats. High incidences of organisms in clothing, luggage etc. have been reported in a number of studies in Antarctica (Hughes & Convey 2010, Hughes et al. 2010, 2011, McNeill et al. 2011). If this is the case, fewer changes in species composition from plot to plot would be found in the anthropogenically influenced areas (lower ß-diversity) than in the non-influenced areas.

The statistical analyses could not generally confirm this hypothesis. There was no general tendency for any animal group to show higher species turnover in the non-influenced areas than in the influenced areas. This could be due to the fact that humans simply do not spread species within a locality. On the other hand, it could also be due to the fact that, since generally similar substrates were sampled within a locality, even the non-influenced microhabitats were not different enough to show large degrees of species turnover. A further difficulty in detecting differences in ß-diversity is the high temporal variability of the soil fauna within the same plots (see, e.g., Goddard 1979a, Ayres et al. 2007). Thus, although differences in ß-diversity may be present, it is possible that they were simply not found during the study years. Nonetheless, some animal groups (i.e., Actinedida and Tardigrada) did indeed show a statistical interaction between anthropogenic influence and locality. Although non-significant, the CAP analyses further generally indicated reduced levels of change in species composition in the anthropogenically influenced areas. Altogether, this means that a reduction in ß-diversity could be ascertained for these animal groups in some localities. Analyzing specific localities (data not shown) could prove that, for these two animal groups, a reduction in ß-diversity in the human influenced plots was found in about 40% of the localities. This percentage is more than would be found by chance, indicating an anthropogenic cause. There was no evidence for a common factor among the localities showing or not showing reduced ß-diversity, such as type of soil substrate, vegetation or wildlife. Thus the reasons why a human influence on ß-diversity was found in some localities and not others cannot be determined. Although not widespread in the localities or animal groups studied, there is therefore indeed some indication of an anthropogenically induced reduction of ß-diversity. Especially considering wildlife, reduced ß-diversity in a few localities exhibiting high penguin or seal populations is remarkable. It would be expected that wildlife constantly travelling from the sea to land and back would also influence the soil fauna's ß-diversity. Thus, in these localities, humans obviously influenced the ß-diversity even further.

Although not consistent among all animal groups, study years or localities, all of the analyses described above show important changes in the soil fauna due to human influence. This indication has disturbing implications for changes in the biodiversity of the soil fauna in the entire maritime Antarctic. It is not surprising that the results were somewhat inconsistent. The localities studied have been experiencing human pressure for many years, if not decades. Thus even the non-influenced areas sampled here probably experience some degree of human influence throughout the years, albeit minor in comparison to the influenced areas. Furthermore, many of the localities experience high wildlife pressure, with large colonies of penguins or seals throughout the Antarctic summer, which themselves will have a major impact on the soils and soil fauna. Thus, all of the anthropogenic effects found here are, firstly, stronger in sites more intensely visited by humans and, secondly, are above and beyond that caused by wildlife. This indicates how strong human influence on the Antarctic soil fauna actually is, despite the somewhat inconsistent results found here, and warrants concern for major negative changes in sites strongly visited by tourists and research personnel.

4.2.3 Introduction of non-native species

Introduced non-native species threaten many Antarctic ecosystems. The inherent species poverty of native Antarctic biotic communities, which are probably unsaturated in species composition, renders them particularly vulnerable to introduced species. A most spectacular case of introduced soil species is the introduction of non-native chironomid (Diptera) and enchytraeid species to Signy Island during experiments in the 1960s. These species have since become established on the island, have increased their populations and the chironomid *E. murphyi* may actively be spreading its distribution (Burn 1982, Block et al. 1984a, Hughes et al. 2010). The lack of functional redundancy in Antarctic soil communities further carries the danger of introduced species irreversibly changing the fragile ecosystems (Hughes et al. 2010, Greenslade & Convey 2012). For instance, the introduction of a non-native isopod species (*P. scabor*) on the Subantarctic Marion Island is expected to alter dramatically decomposition processes of organic matter in soils (Slabber & Chown 2002).

Hughes et al. (2010) distinguish three different distributional types of species: (1) endemic species limited to Antarctica or even specific Antarctic regions (probably pre-glaciation relics), (2) native Antarctic species that are most likely post-glaciation colonists and are generally cosmopolitan in distribution and (3) (usually human) introduced non-native species. Greenslade & Convey (2012) elaborate upon this, discriminating between “endemic”, “native” and “naturalized” species, the latter having established viable, reproducing populations after their introduction. They further differentiate between “exotic” (= non-native to Antarctica), “introduced” (usually by humans) and “invasive species”, which actively spread their distribution beyond their point of introduction. Up to now five non-native (= exotic) species have provenly established populations in the maritime Antarctic (1 chironomid, 1 enchytraeid and 3 *Poa* species), although evidence exists that other taxa successfully colonize terrestrial maritime and continental Antarctic habitats (Hughes et al. 2010, Chown et al. 2012). The present study primarily concerns the identification of introduced, non-native (= exotic) species in the sites studied.

An abrupt reduction in the diversity and species richness of the invertebrate fauna exists between the Subantarctic and maritime Antarctic zones south of the Antarctic Convergence. This is considered to be due to barriers to colonization by non-native species (Block 1984a), of

which various types have been postulated (Hughes et al. 2010): (1) an immigration or transport barrier due to the isolation of Antarctic habitats, (2) an establishment barrier due to extreme environment conditions preventing survival of colonizing species, although the present climatic changes are expected to lower this barrier (Convey 2006), (3) an invasion barrier limiting the spread of surviving non-native species and finally (4) a transformation barrier which prevents changes of an invaded ecosystem. Many of these barriers are a result of the energy and resource limitations of Antarctic ecosystems, which act as filters to colonization, establishment and development of non-native species (Ellis-Evans & Walton 1990). Despite these manifold barriers, the number of proven cases of species introductions into (Sub)Antarctic habitats from more northern locations has increased in recent years (Convey & Stevens 2007). For instance, Pugh (1994) lists 70 of 520 species of mites in Antarctica and Subantarctica as possibly originating from other continents, especially Australia, South America and Europe.

The most important limitations to invertebrate colonization are assumed to be the geographical isolation of Antarctic habitats (Block 1984a). Various mechanisms of natural species dispersal into these isolated habitats have been considered in the past: wind dispersal, via the feathers or feet of birds and "rafting" on natural debris and logs in sea currents (Gressitt & Yoshimoto 1963, Ellis-Evans & Walton 1990, Pugh 1994, Marshall & Pugh 1996, Hogg & Stevens 2002, Hughes & Convey 2012).

In a number of studies, short-range dispersal between Antarctic habitats by wind has been proven or is considered to be the only possible dispersal mechanism, e.g., in the colonization of glaciers or isolated nunataks etc. (e.g., Gressitt 1967, Tilbrook 1967b, Hawes et al. 2007, Block 1979). Long-range dispersal of non-native species by wind into Antarctic habitats, however, is assumed to be highly unlikely or extremely sporadic due to desiccation of the individuals in dry air, against which most soil invertebrate species are very vulnerable (Gressitt & Yoshimoto 1963, Schatz 1991, Pugh 1994, Marshall & Pugh 1996). A possible exception may be infrequent major storms with anomalous wind patterns, which can carry large amounts of biological material as well as pass over southern South America, the Subantarctic islands as well as major parts of the Antarctic continent in as little as eight days (Gressitt & Yoshimoto 1963, Ellis-Evans & Walton 1990).

Although little proven evidence for the transportation of soil invertebrates by birds exists (Gressitt & Yoshimoto 1963, but see Strong 1967 for individuals of *Alaskozetes* [Oribatida] found in feathers of skua), some authors consider this a true possibility, especially for inundation-intolerant mites (e.g., Actinedida) or in the northernmost maritime Antarctic areas (e.g., the South Sandwich Islands) (Pugh 1994, Convey et al. 2000a). Two other general mechanisms of transportation of invertebrates by vertebrate animals exist: parasitism and phoresy (= dispersal of transportation-specific life-cycle stages of soil animals via other larger animals). Essentially only nasal parasites of seals and birds being briefly set free are known from the Antarctic and Subantarctic (Pugh 1997) and may account for the sporadic *Lorryia* s.l. (Actinedida) individuals found in the present study. Phoresy is also relatively unknown in Antarctic habitats, and has only been considered for species of the family Tarsonemidae (Actinedida) (Pugh 1997). Such bird-mediated dispersal may thus be the source of the sporadic Tarsonemid individuals found in the present study, often in the vicinity of penguin rookeries.

Transportation by driftwood etc. ("rafting") has also been assumed to be very unlikely as a distribution mechanism (Gressitt & Yoshimoto 1963, Pugh 1994), since salt water is highly unfavourable to almost all invertebrates. Although this essentially eliminates active swimming,

rafting has been experimentally tested and found to be possible (Strong 1967, where the oribatid *Alaskozetes* could survive for more than 50 days) or by oribatid mites having been found alive in clefts of driftwood found on shores (Schatz 1991). Thus, rafting is assumed to be a possible dispersal mechanism for, e.g., inundation-tolerant mites such as Gamasina and Oribatida (Convey et al. 2000a). Convey et al. (2000a) consider this to be an explanation for the acarine fauna of the South Sandwich Islands being related to the fauna of South Georgia, the South Orkney Islands etc. Long-range dispersal by rafting into more southern areas is nonetheless unlikely, since the Antarctic circumpolar current flows act as an isolating mechanism.

Although many of these natural dispersal mechanisms are indeed possible or even likely, they are probably mostly limited to short-range dispersal. Thus, the most effective dispersal agent of non-native species into Subantarctic and Antarctic habitats is probably humans (Block 1984a, Pugh 1994, 1997, Frenot et al 2005, Greenslade & Convey 2012). The most common human-mediated transportation of non-native species into Antarctica has been through living organisms or propagules in imported foods, domestic animals, plants, ship stores etc. (Gressitt & Yoshimoto 1963, Schatz 1991, Pugh 1994, Greenslade & Convey 2012, Hughes et al. 2010). Species from various taxonomical groups can be easily transported into Antarctica in soil, e.g. attached to plants, vehicles etc. and remain viable (Pugh 1994). Hughes et al. (2010) report on over 100 kg of soil on vehicles brought to one research station, which contained tens of thousands of seeds as well as bacteria, fungi and numerous invertebrate animals, including 11 non-native nematodes species. The increasing number of tourists in Antarctica also bears the danger of transporting non-native taxa with their clothing and luggage (Whinam et al. 2005, Hughes & Convey 2010, Hughes et al. 2010, 2011, McNeill et al. 2011, Hughes & Convey 2012). Despite the large number of tourists, the largest risk of transfer of non-native species per visitor is associated with scientists, station personnel and tourist support (expedition) personnel (Chown et al. 2012). The vast majority of such personnel have previously travelled to other cold climate areas, therefore increasing the inherent danger of transferring non-native species capable of surviving extreme conditions. Greenslade & Convey 2012 consider the recent increase in records of non-native species not to be an artefact of increase collecting, but rather due to increasing human activities.

Due to humans being the largest vector of non-native species, many such taxa have been confined to human habitations in the Antarctic (Rounsevell 1978, Vogel & Nicolai 1983, Block 1984a, Pugh 1994, Greenslade & Convey 2012), although some taxa are known to be increasing in distribution, especially in Subantarctic islands. Since most humans visiting and working in Antarctica are Europeans, both historically as well as in recent times, the vast majority of non-native species are either European in origin or cosmopolitan with a large ecological range (Gressitt & Yoshimoto 1963, Crafford 1986, Pugh 1994, Frenot et al 2005, Greenslade & Convey 2012).

If the dispersal barrier has been overcome, non-native species must then establish reproducing populations before they become a threat to Antarctic habitats. In other words, the establishment barrier must be overcome. Convey et al. (2000a), for instance, report that Subantarctic species frequently arrive on the South Sandwich Islands, but are unable to establish viable populations (*Cryptopygus caecus* being an exception). For successful colonization, various species characteristics are required: e.g. an acceptable new microhabitat, physiological adaptations and a successful reproduction strategy (Ellis-Evans & Walton 1990,

Greenslade & Convey 2012). The most often cited necessary physiological adaptation is, of course, cold tolerance (Burn 1982, Block 1984a, Ellis-Evans & Walton 1990, Pugh 1997, Hughes et al. 2010, Chown et al. 2012). Certain life-cycle characteristics are equally important, one of which being the ability to complete a generation cycle under Antarctic conditions (Block 1984a), especially via a lengthened larval or nymph cycle (Crafford 1986, Pugh 1994, Frenot et al 2005). Although sexual reproduction is predominant among native Antarctic species, a most important attribute of non-native species in Antarctica is parthenogenetic reproduction (Crafford 1986, Ellis-Evans & Walton 1990, Frenot et al 2005, Greenslade & Convey 2012). Most established non-native species possess these physiological adaptations, so that these characteristics are usually considered to be pre-adaptations necessary for colonization (Block 1984a, Crafford 1986, Ellis-Evans & Walton 1990, Pugh 1994, Hughes et al. 2010).

For the establishment barrier to be overcome, certain habitat characteristics are also necessary for successful colonization by non-native species. The most obvious for terrestrial soil fauna is the necessity of ice-free locations and periods of favourable climatic conditions (Ellis-Evans & Walton 1990, Hughes et al. 2010). In the studied region, these generally coincide with the areas and times of highest human (usually tourist) activities, so that non-native species introduced by humans are commonly brought to the most favourable sites for their establishment. In greatest danger of introduction and establishment of non-native species is therefore the western Antarctic Peninsula, which has the mildest climatic conditions, the most ice-free areas in Antarctica as well as the highest tourist pressure (Chown et al. 2012). Other factors include the availability of free ecological niches as well as few competitors or predators (Ellis-Evans & Walton 1990, Schatz 1991, Frenot et al 2005). Important is furthermore not only the availability of moist soils, but especially nutrient availability (Ellis-Evans & Walton 1990, Frenot et al 2005, Caruso & Bargagli 2007, Greenslade & Convey 2012). Even relatively barren mineral soils in Antarctic habitats contain substantial quantities of biologically available organic molecules, which are available to microorganisms that in turn serve as nutrient resources for protozoan and invertebrate animals. One of the most important correlates of introduced non-native species are the availability of barren or disturbed soils, the rough textures of which promote primary colonization of new species (Ellis-Evans & Walton 1990, Chown et al. 2012, Greenslade & Convey 2012).

In the evaluation of whether a previously unrecorded species is non-native or not, all the factors mentioned above must be considered. In a recent article, Hughes & Convey (2012) listed a number of criteria for such evaluations, such as fossil evidence, historical evidence, habitat characteristics, geographical distribution of the species, frequency of known naturalization, genetic diversity, reproductive patterns as well as possible means of introduction. The present study could not provide the data necessary to evaluate all of these criteria. Thus the determination of a previously unrecorded species found in the present study as non-native was necessarily limited to the following criteria:

- previously unrecorded in Antarctic habitats,
- known distribution as cosmopolitan or European (no South African, South American or Australian taxa were found in the present study) and
- life-cycle or habitat characteristics, such as parthenogenesis or larger densities found in barren soils (i.e., with no vegetational cover).

Direct evidence of species being introduced by humans was not found in the current investigation. Therefore recorded non-native species are only considered to be *potentially* introduced if they were found in areas with high human pressure.

Although many previously unrecorded taxa were found in the present study, these criteria do not allow many of these to be definitively listed as non-native and potentially introduced. This is particularly the case with the Nematoda and Tardigrada. Approximately 85% of the nematode species known from Antarctica are endemic to the area. Potentially introduced species can only be possibly found among the remaining 15%, which consists of eight species known to occur in Antarctica, but having a broad, at times cosmopolitan distribution: *Eumonhystera vulgaris, Geomonhystera villosa, Ceratoplectus armatus, Coomansus gerlachei, Pratylenchus andinus, Tylenchorhynchus maximus, Aglenchus agricola* and *Paratylenchus nanus* (Maslen & Convey 2006, Ryss et al. 2005). The first four of these were also found in the present investigation. Further species were identified in this study (*Aporcelaimellus* cf. *obtusicaudatus, Cervidellus* cf. *vexilliger, Pelodera* cf. *strongyloides, Pellioditis* cf. *marina* and *Rhomborhabditis* cf. *teres*) which represent first records for Antarctica and are morphologically very similar to cosmopolitan species. However, it remains open whether these specimens can truly be classified as the corresponding widely distributed species or whether they represent only related, morphologically similar species with a limited distribution in Antarctica. These questions can only be verified in future comparative taxonomic studies, preferably on a molecular biological basis.

As mentioned above (Section 4.1.2), compared to other parts of the world, the nematode fauna of Antarctica has been fairly well studied in the past, although often with other substrates as those sampled in the current investigations. The fact that 19 of the 40 nematode species recorded in the present study have not been previously reported from Antarctica can therefore not be regarded as an indication of a recent colonization by these species or even for an introduction of non-native species. It is rather quite possible that these species have simply been overlooked in the past, due to their small-scale very heterogeneous occurrence and the limited number of detailed studies on soil-living nematode communities.

Even for species exhibiting a global distribution and thus having potentially colonized Antarctica at some point (provided they did not originate there and subsequently spread throughout the world), possible immigration pathways have not at all been clarified. One possibility was reported by Spaull (1973), who found *Caenorhabditis* sp. and another unidentified representative of rhabditid nematodes in dried mud washed from the feet of seabirds (Sheathbills: *Chionis* spp.), which could thus obviously act as a transportation medium for nematodes between Patagonia and Antarctica. Among those nematode species recorded here for the first time from Antarctica, three species also belong to the rhabditids: *Pelodera* cf. *strongyloides, Pellioditis* cf. *marina* and *Rhomborhabditis* cf. *teres*. These species are bacterial feeders, which can develop large populations within weeks provided a sufficient nutrient supply is available, as was the case in the ornithogenic soils on Paulet Island. On the other hand, they can also develop dormant stages (formation of a non-feeding “dauer larva” or quiescent to cryptobiotic states; see above) as a result of adverse environmental conditions or reduced food resources. In these stages, such species can not only survive such conditions, but are also well protected during passive transportation, be it in the feathers or feet of birds or in the soles of people’s boots.

Such dormant stages can play a role during passive wind transport, especially regarding survival of low temperatures and humidities at high altitudes. Nkem et al. (2006) studied wind dispersal and found three times as many inactive as active Nematodes in the air plankton. Spaull (1973), on the other hand, considers wind transport of South African or South American nematodes very unlikely, since the predominant circumpolar winds in Antarctica generally block a north to south movement between South African or South America and Antarctica (with the exception of major storms, see above). The very different species composition of the nematode faunas on the various islands, even those with a similar vegetation, further indicate such and other small-scale dispersal barriers.

No indication of human transport of Tardigrada was found. While the statistical evidence does not conclusively show that tourism has a major impact on total tardigrade populations, e.g. transporting species, there has only been speculation on this subject with no previous tardigrade studies with which to compare the data. All the tardigrades recorded in the present study are local to the Antarctic, with most being endemic to Maritime Antarctica and some with even narrower distributional ranges. Only one species, *Calohypsibius* sp. (previously reported as *Calohypsibius* cf. *ornatus*) had not been previously recorded south of South Georgia. Although the individual numbers of this species were very small, it was found in an anthropogenically non-influenced site, suggesting that this record is more an effect of a lack of previous knowledge than human intervention. Several potentially new species were recorded in this study (e.g., *Isohypsibius* sp., *Diphascon* (*Adropion*) sp.), but - as the initial taxonomic analysis shows - these cannot be determined to known species, indicating potential endemism to the maritime Antarctic rather than non-native imports.

Among the mesofauna, no species of Gamasina or Oribatida were determined that could be suspected of being non-native. However, this conclusion cannot be drawn for those oribatid taxa that could not be definitively determined to species level, (e.g., *Brachychochthonius* sp. and *Liochthonius.* cf. *mollis*). An introduction of non-native Oribatida in Antarctica is, however, indeed possible and has been described in the literature. Pugh (1994) listed seven species of European and South American origin that are probably non-native, six from a Subantarctic and one from an Antarctic location (*Gressitoppia pepitensis* (Hammer, 1962) from Greenwich Island, South Shetland Islands). This author suspected agricultural products, soil and leaves on imported plants as well as bark from lumber as vectors of these species.

Based on the criteria listed above, convincing evidence for non-native species recorded in the present study was therefore only found for Collembola and Actinedida. The highest number of non-native species of these taxon groups was found on Deception Island. While no non-native actinedid species were previously known on the island, the present study identified four potential species: *Alicorhagia* sp., *Coccotydaeolus krantzii*, *Speleorchestes* sp. and *Terpnacarus glebulentus*, none of which have been previously recorded in the Antarctic at all. All of these taxa are known from different areas in various continents (see Results section for their distributions) and thus may be considered to be cosmopolitan. As opposed to all native species, *C. krantzii* was also found in significantly higher densities in barren soils (e.g., with no vegetation). These two characteristics further indicate these species' status as non-native.

Four non-native collembolan species were previously known to inhabit Deception Island (*Hypogastrura viatica, Folsomia candida, Protaphorura fimata, Cryptopygus caecus*) (Greenslade & Wise 1984; Greenslade 2010), the highest recorded for any location in the maritime or continental Antarctic. This is only paralleled or exceeded by some Subantarctic islands (South

Georgia [4 species], Macquarie Island [11], Marion Island [5], Kerguelen Island [6], Crozet Island [3]; Greenslade and Convey 2011). Three additional non-native Collembolen species were recorded on Deception Island during the current project (including simultaneous collections by the British Antarctic Survey [BAS]): *Mesaphorura macrochaeta*, *Proisotoma minuta* and *Deuteraphorura cebennaria*. *M. macrochaeta* was not present in the BAS collections obtained from the vicinity of Whaler's Bay, indicating that it is likely to currently have a very restricted distribution in this area. It inhabits a large range of habitats from forests to arable fields and is widespread in temperate climatic zones. *M. macrochaeta* has spread to high latitudes in the Arctic (Fjellberg 1994; Babenko & Fjellberg 2006) and as far south as Macquarie Island in the Subantarctic (Greenslade 1992). It was probably originally introduced to the Southern Hemisphere in imported soil and peat moss (Greenslade 2006). In the samples of this study, *M. macrochaeta* was found in significantly higher densities in barren soils (i.e., with no vegetation), which is considered to be a characteristic of non-native species. In the present study, *P. minuta* was found at Caliente Hill from a warmed area (BAS collection) and a single specimen from a trampled site at Whalers Bay. This species is also cosmopolitan in distribution and is normally found in habitats with high organic matter content. It is more frequent in southern areas, including the tropics (Potapov 2001), having become naturalised on Macquarie Island, where it is thought to have been introduced in peat moss along with *M. macrochaeta* (Greenslade 2006). *D. cebennaria* was present in one sample obtained from a *Polytrichum alpinum* patch at Pendulum Cove. This area was heavily impacted by the 1968 eruptions (and thus represents disturbed soils) and is a popular tourist visitation site.

The current study thus brings the total known collembolan species from Deception Island to 14, including seven non-native species (if *Cryptopygus caecus* is considered to be non-native), thus representing half of the total collembolan fauna there. This high number of non-native species can be attributed firstly to warmer and moister conditions due to geothermal activity. Deception Island consists of an active caldera, with the last major eruptions occurring between 1967 and 1970 (Baker et al. 1975). The active volcanic nature of the island with various types of geothermally influenced habitats (Smellie et al. 2002) cause its terrestrial biology to be exceptional for the Antarctic (Smith 2005a, 2005b, Convey & Smith 2006). Geothermally warmed habitats, which are only found in the maritime Antarctic on Deception Island and in the South Sandwich Islands, host a range of native plant and invertebrate species otherwise unknown in the Antarctic (Aptroot & van der Knaap 1993, Convey et al. 2000, Convey & Smith 2006). The records from Deception Island are consistent with some conclusions of Gabriel et al. (2001) that the likelihood of a community being invaded depends, at least partly, on temperature. In contrast, Terauds et al. (2011) found no difference in the spatial distribution of non-native and indigenous species with environmental factors on the Subantarctic Macquarie Island, although their analysis did not take into account the different biologies and biogeographic affinities of native species. Greenslade (2006) reported that the collembolan fauna at higher altitudes included no non-indigenous species. In contrast, coastal sites harboured every non-native species recorded from the island, and the native species found there have affinities with New Zealand's southern islands to the east. This further indicates transportation pathways leading mostly to coastal areas, corroborating this and previous studies showing the highest number of non-native species in the western maritime Antarctic.

The long history of human presence on Deception Island and the current high level of tourism have also been proposed as underlying the colonization of the island by non-native species

(Downie et al. 2000). The island currently experiences one of the highest visitation rates in the Antarctic by cruise ships and tourist yachts, although focused on a limited number of sites (e.g., Whalers Bay with more than 16,000 tourists reported to have landed in 2009–2010 [Lynch et al. 2010]). Five of seven non-indigenous collembolan species (*Mesaphorura macrochaeta, Proisotoma minuta, Folsomia candida, Protaphorura fimata, Deuteraphorura cebennaria*) were recorded only in areas with considerable human influence. They were identified as being of the highest introduction-risk status (Greenslade & Convey 2011). It is assumed here, however, that these species belong to the moderate risk-status group, since thus far they were found only in warmed sites of Deception Island, but not in other parts of the maritime Antarctic (see below).

Hypogastura viatica is one of the best known non-native microarthropod species in the maritime Antarctic. It is cosmopolitan, first described from Sweden. The species is frequent in Northern Europe, including the Arctic (Fjellberg, 1998). In the Southern Hemisphere, *H. viatica* is considered to be an exotic species in the Antarctic and Subantarctic (Wise 1967, Greenslade, 2010, 2006, Convey et al. 1999) due to unpredictably high local densities, penetration into and predomination in several inland biotopes, a preference for disturbed and organically enriched sites and its cosmopolitan distribution. Hack (1949) first recorded *H. viatica* on Deception Island. Wise (1971) later recorded the species from Tower Island between Deception Island and the coast of Antarctic Peninsula. Greenslade (1995) extended its distribution much further south to Leonie Island near Adelaide Island (ca. 67°36′S). Apart from *H. viatica*, no non-native Collembolan species has been previously recorded from any other location in the maritime Antarctic beyond Deception Island, which is confirmed in the present study. *H. viatica*, on the other hand, was recorded in five locations in the present study. Three new records of this species during the current project are from Halfmoon Island in the South Shetland Islands, Neko Harbour on Graham Coast and Devil Island, all regular tourist visitation sites. This species is thus not restricted to warmed areas of Deception Island, but is widely distributed throughout the region.

The biology of each species is also likely to have played a role in their ability to colonise Antarctic habitats. In this respect, it is significant that three of the non-indigenous collembolan species are parthenogenetic (*Mesaphorura macrochaeta, Protaphorura fimata, Folsomia candida*). Although not definitively known for the potentially non-native actinedid species, males are not known for these taxa so that they are also most likely parthenogenetic. Chahartaghi et al. (2009) have demonstrated that parthenogenetic species of Collembola colonise vacant areas more quickly than those capable of sexual reproduction. *Hypogastura viatica* is not parthenogenetic, but is known to have established and become invasive on several Subantarctic islands, and has been recorded from elsewhere in the maritime Antarctic (Greenslade 1995, Frenot et al. 2005, Greenslade & Convey 2011; see below). It appears able to outcompete the native *Cryptopygus antarcticus antarcticus* in coastal sites on South Georgia (Convey et al. 1999). The same may possibly occur in Deception Island, but further studies are necessary to prove this.

According to the data provided here, *Hypogastrura viatica* shows contrasting densities in different islands of the maritime Antarctic. During the two years of the present study, more than 11,000 individuals were recorded from Deception Island and up to several tens on Devil Island, Neko Harbour and Halfmoon Island. Wise supposes (after Greenslade, 2010) that *H. viatica* was introduced and only invaded the region in the middle of the 20th century. Our data

confirm this suggestion. The species is probably experiencing positive expansion dynamics on Deception Island.

Table 20: Densities of potentially non-native species found in the localities of the present study. All values in individuals per 100 cm^3 substrate. Species marked with an * were recorded for the first time in Antarctica during the current study.

		Collembola				Actinedida (Acari)				
Locality	**Region**	*Hypogastrura viatica*	*Mesaphorura macrochaeta* *	*Cryptopygus caecus*	*Proisotoma minuta* *	*Speleorchestes* sp.*	*Alicorhagia* sp.*	*Coccotydaeolus krantzii**	*Terpnacarus glebulentus**	**Total species number**
Arctowski Station (2010)	King George Island					0.5				1
Biologenbucht (2010)						0.1			0.2	2
Punta Cristian (2010)						0.3			0.1	2
Punta Cristian II (2010)								0.1	0.1	2
Ardley Island (2011)										0
Halfmoon Island (2010)	Livingston Island	0.2				0.1				2
Halfmoon Island (2011)		1.1								1
Hannah Point (2011)								0.1		1
Whaler's Bay (2010)	Deception Island	551	1.3	1.3	0.1	0.4	0.1	0.4		7
Whaler's Bay (2011)		206	0.1	0.2		3.8		65.8	1.2	6
Telefon Bay (2011)		0.4								1
Petermann Island (2010)	Antarctic Peninsula					0.2		0.2		2
Neko Harbour (2010)		0.1						0.2		2
Neko Harbour (2011)		2.5				0.1		0.6	0.1	4
Devil Island (2010)	Weddel Sea	0.7								1
Paulet Isand (2011)										0

Most of the non-native actinedid species registered in the present study were also found in a number of sites beyond Deception Island, including those on King George Island (Table 20). Although individual numbers of non-native species in these sites were small, all of the sites have a relatively high human presence. While is not yet certain that the individuals of, e.g., *Speleorchestes* represent only one species, the specific species status of the other taxa indicate a possible spread of these species among sites with high human pressure. The second highest quantity of potentially non-native species as well as their densities was found on Neko Harbour. This location also receives one of the highest number of tourists per year, substantiating the indication of a potential spread of these actinedid species due to human influence. Therefore, as in *H. viatica*, these species may also be becoming invasive and should be given a high introduction-risk status.

Thus far *Hypogastrura viatica* has not been found on King George Island, which is climatically favourable and thus appears highly likely to receive this species. This island has been subject to

intensive investigation of Collembola during the last decades (Gressitt et al. 1967, Wise 1971, Usher & Edwards 1986, Bulaintsev 1990, Greenslade 1995, 2010, Ohyama & Shimada 1998, Yue & Tamura 2001), resulting in its fauna being well known. It can therefore be postulated that up to now this species is absent or has a minimal occurrence in the relatively species-rich collembolan communities of King George Island.

As opposed to most other non-native species in the present study (see above), *H. viatica* was found in its highest abundances under middle levels of vegetational cover, further indicating its expansive tendencies. Furthermore, according to the present data, this species attains very high abundances in coastal sites of Whalers Bay, contrasting with the apparent low density and few individuals reported there in the past (Wise 1967, 1971) and with new records from Collins Point and Caliente Hill (BAS samples). This affinity for coastal areas is possibly shared by another probable non-native Collembolan, *Cryptopygus caecus*, which, however, appears to be limited to warmed sites in the maritime Antarctic. Almost all non-native actinedid and collembolan species recorded here showed no significant difference in abundances between touristically influenced and non-influenced coastal sites, further indicating the potential for being distributed through human activities.

4.3 Efficiency of the bootwashing methods used on the MS Hanseatic

A shore leave to a Subantarctic site was intentionally chosen for an examination of the bootwashing methods aboard the MS Hanseatic, since a stronger biological activity in the soils and thus a larger potential for unintentional human transport of organisms was expected than in Antarctic locations. After visiting the Subantarctic South Georgia, the inspected footwear of the passengers was considerably more contaminated with soil organisms than after visiting Antarctic locations. This clearly illustrates the danger of unintentionally transporting non-native species from the Subantarctic to Antarctica.

The strong growth and activity of microorganisms in the boot samples after visiting South Georgia - despite the use of a disinfection agent and conserving the samples in 50% ethanol - is alarming. Although transfer of bacteria is natural and unavoidable, microbial activity to this extent is unusual. The microbial activity was most likely fostered by penguin droppings on the passengers' boots and exemplifies the biological vitality of the soil substrates attached to human footwear. The aqueous solution and the dissolved nutrients as well as the time period of four weeks between sampling and examination of the samples certainly offer the microorganisms optimal conditions and sufficient time for growth. Due to the much shorter time periods between land visits by cruise ship passengers, the microbial growth conditions will be clearly poorer. Nonetheless these results clearly illustrate the high biological potential of soils attached to footwear and the necessity of preventive measures such as "bootwashing".

Although the replicate numbers of the examined passengers' boots was fairly small, an unexpected number of plant remains and soil animals was detected. In the few samples, at least one nematode per land excursion was identified with certainty. The transportation of soil organisms on the footwear of visitors to Antarctica was therefore proven. Due to the conservation of the samples, it is not known whether the detected individuals were alive and still viable after being transferred by the passengers. Therefore an actual transfer of biologically active organisms was not demonstrated. Conserving the samples also did not allow the efficacy of the disinfectant and the duration of its application to be evaluated. Further ecotoxicological experiments are recommended for these questions. At least the microorganisms (bacteria etc.)

remained alive after one pass through the disinfectant. Despite all these uncertainties, these results document that the potential of a transfer of soil organisms from locality to locality undoubtedly exists.

Due to the low number of samples in the boot inspections, an exact quantification (e.g., in percent) of the efficiency of the washing methods of the MS Hanseatic could not be performed. Merely the existence of soil organisms despite the cleaning measures does not attest to deficient efficiency of the washing measures and the disinfectant. In fact, human behavior could more likely be responsible for the transfer of soil organisms. The passengers on board the cruise ship were insufficiently informed about the necessity of cleaning the footwear or about the operation of the bootwashing equipment. Only a few of the passengers were acquainted with the measures through the information provided with their travel documents; many passengers obviously did not read this. All of these factors led to an insufficiently efficient use of the equipment. This in turn illustrates the necessity of such prevention measures being guided and supervised, e.g., by ship crew.

Commendatory is the fact that, after land excursions and localities of the Antarctic Peninsula, ship crew hand cleaned the footwear and pants of all passengers *before* they returned to the ship. This cleaning was extensive, thorough and carried out with visual control. The success of this measure could be observed during the control washing of passengers boots afterwards, as the samples were clearly cleaner after this handwashing. Much less sediment and soil organisms were present in the samples than those that were not previously hand washed. Microbial activity was rarely present, which is also most likely a result of this pre-cleaning. However, this measure was carried out only in a few localities along the Antarctic Peninsula.

4.4 Recommendations

Based on the results of the present project, the following recommendations concerning a limitation of anthropogenic impacts on terrestrial Antarctic ecosystems can be given:

- **Intensification of biosecurity measures to prohibit introduction of non-native species**

The Environmental Protocol to the Antarctic Treaty contains provisions that aim at limiting or prohibiting intentional introductions of non-native species into Antarctic ecosystems. There is little mention of unintentional introductions or movement of species within Antarctic habitats (Hughes et al. 2010). The accidental introduction of non-native species is, however, one of the largest dangers to Antarctic ecosystems. To improve and intensify biosecurity instruments, different measures are necessary:

  - **Intensification of (on-board) tourist education; better control of use of biosecurity measures**

Even the best procedures to prohibit the transfer of non-native species will be breached through ignorance (Greenslade & Convey 2012). It is therefore necessary that all tourists and research personnel be thoroughly briefed on the necessity and use of biosecurity measures before entering Antarctic waters. This proved to be lacking on the cruise ship observed in this study and necessitates improvement. A stronger control of the correct use of biosecurity measures by tourists should also be performed by the responsible ship personnel. Biosecurity methods must be employed on all ships (tourist as well as research) and international regulations should be imposed. Previous studies could further show that especially scientists

carry a greater propagule load than tourists and that the largest propagule transfer per visitor is associated with science-program and tourist-support personnel, who often visit Arctic habitats before travelling to the Antarctic (Chown et al. 2012). While ship-issued backpacks and boots may reduce propagule pressure from tourists, research and tourist support personnel often use their own equipment. Therefore, precautionary measures must also be employed by these groups, which should be subject to higher intensity biosecurity methods.

- Intensification between Subantarctic and Antarctic sites

Biosecurity measures are often only employed once Antarctic waters have been entered. However, pre-adapted non-native species are abundantly found in Subantarctic habitats. The present study could show high levels of propagule transport after visiting Subantarctic sites. It is highly important that Subantarctic islands do not act as stepping stones for transport of non-native species to Antarctica (Greenslade & Convey 2012). Therefore, heightened biosecurity measures should be particularly employed between the two areas.

- Intensification after high-risk sites (e.g., Deception Island, Neko Harbour)

"High risk" sites can be considered those with many established non-native species. This is the case for Deception Island, which is strongly enriched with non-native species both at a stage of initial introduction and expansion. The present study could also identify Neko Harbour as a high risk site. Precautionary measures must be stronger after visiting such sites – e.g., carried out before reboarding a ship – to prevent a further spread of these species. *Hypogastrura viatica* has been identified as having the highest risk of invading further Antarctic ecosystems. Its preference for coastal sites renders this species open to transfer by tourists visiting coastal areas. Thus, other localities harbouring this species should also be considered to be high risk sites.

- Stronger protection of microhabitats (increase of "no-go areas")

"No go areas" and ASPAs have usually been determined based on large colonies of wildlife or a closed vegetation cover. The present study could show that the soil fauna significantly increased in abundance and diversity even under lower levels of vegetational cover. Such areas are usually ignored by, e.g., expedition teams leading tourists. Importantly, in locations with only sporadic vegetational cover, human impact was significantly more negative than if no and/or much vegetation was present. Indigenous soil faunal communities also react positively to parameters related with organic matter and water content. During touristic and research visits, areas with initial or sparse vegetation as well as melt-water streams etc. must be more strongly protected. More such areas should be included in "closed area" categories and associated recommendations should be developed and given to tourists and scientists visiting such areas. A stronger education of the ecological importance of such microhabitats is necessary for tourist- and research-support crew.

- No further increase of tourist visitation sites

The potential spread of non-native species throughout touristically visited sites indicated in the present study is alarming. Furthermore, some form of negative anthropogenic impact on soil fauna was found in almost every locality studied. It is therefore to be expected that such impacts would continue in a cumulative manner, especially in light of increasing tourism in Antarctica. In order to stop further expansion of non-native species and to limit further human impacts, a stronger regulation of the number of sites visited, e.g., by tourist cruise ships, is

necessary. Stronger protection of pristine sites without wildlife (penguins, seals and seabirds) is also warranted. In order to contain human impacts - which will definitely continue in the future – a “positive list” of potential visitation sites is called for, outside of which visits by humans should be forbidden. A “positive list” must necessarily be the subject of international agreements and regulations.

- **Regular monitoring**

The present study in combination with previous investigations has been of exceptional importance in creating a baseline of information for the soil fauna present in Antarctic soils, from which further studies can radiate. Long-term monitoring of all established sampling sites would offer a better understanding of how annual human visitation (research and tourism) affects soil organisms through time. Such monitoring should also explore in greater clarity the potential for direct anthropogenic impact via micro-transport of soil organisms, both inter- and intra-island, on Antarctic tourist routes. Long-term data provided by monitoring programs would greatly enhance the formulation and performance review of existing and future precautionary and mitigation measures. Such monitoring programs must be performed within international programs, be based upon internationally agreed-upon standards and should especially include those groups known to contain non-native species and for which the most widespread data is available (e.g., Collembola, Actinedida).

5 Summary

Human activity in Antarctica has increased enormously in the last two decades. This concerns not only research and logistics, but especially also tourism. The focus of touristic activities is generally concentrated in certain regions of the maritime Antarctic. Although terrestrial soil-dwelling organisms in Antarctica have been relatively well studied in the last 50 years, hardly any studies on human impacts on soil organisms exist. Only in the last years has the introduction of non-native soil-animal and plant species in Subantarctic and Antarctic habitats been specifically investigated. The present study therefore aimed at assessing the impact of human activities on existing soil-organism communities - with a focus on soil-dwelling invertebrate animals - as well as the potential introduction of species not native to Antarctica.

For this purpose, soil organisms were studied in areas strongly frequented by humans. Emphasis was placed on the Fildes Peninsula of King George Island, where a strong concentration of researchers is found, as well as on areas in the northern and western maritime Antarctic along the travel routes of cruise ships. The study aimed at answering the following questions: (1) Have non-native soil-animal species been introduced into areas of frequent human activity? (2) How effective are existing biosecurity measures against the transfer of soil organisms? (3) Does human activity have a direct impact on Antarctic soil-organism communities? (4) Which habitat parameters or interactions between human activity and habitat parameters influence the occurrence of Antarctic soil organisms? (5) Does an anthropogenic transfer of species lead to a "homogenisation" and thus to a reduction in the diversity of the species composition in different Antarctic habitats?

In the Antarctic summers of the years 2009/2010 and 2010/2011, soil organisms were registered and compared in a total of 13 localities. In each locality, soil samples were taken in areas both influenced and not influenced by humans. In the two study years a total of 327 samples were taken and studied. European taxonomic specialists of the taxonomic major groups cryptogam plants, Nematoda, Tardigrada, Collembola as well as the mite groups Actinedida, Oribatida and Gamasina undertook the determination of the recorded plant and soil-animal species. Furthermore, the soil substrates of all samples were submitted to an extensive soil analysis.

A total of 35 Antarctic plant species were determined. Introduced non-native species could not be identified. The botanical surveys - including the registration of vegetational cover and the plant communities - mostly served as background parameters for the soil-zoological studies. The phytosociological results confirmed the comparability of the anthropogenically influenced and non-influenced areas, as did the results of the soil analyses of the sampled substrates.

More than 320,000 individuals from a total of 98 species could be recorded from the studied animal groups. The registered communities are characteristic for maritime Antarctic habitats. Among the Tardigrada and Nematoda, species were recorded that have not yet been found in Antarctica. Their status as native or non-native could not be evaluated, since they are new to science or due to lacking comparative studies in the past. Among the Collembola and Actinedida, eight species were identified as being non-native and potentially introduced. A human-induced dispersal of these species appears to be highly likely. An insufficient efficiency of the biosecurity measures could be discerned. They proved to be especially ineffective in areas that were very muddy or which harboured species- and individual-rich soil-animal

communities. The insufficient efficiency of these measures appeared to not be primarily due to the methods themselves, but rather to poor human behaviour during their implementation.

Concerning an impact of human activities on the soil fauna, the results were often masked by high data variability. However, precise statistical procedures could clearly reveal a significant influence of humans on the studied soil fauna. At the level of the total soil-animal communities, a human influence generally led to reduced individual densities. The reactions of single species to human activities were different: the populations of many species were reduced, while those of other species actually increased. The total densities of most animal groups and especially the densities of many species increased with increasing vegetational cover. The influence of humans was strongest at middle (up to 50%) and higher (up to close to 100%) levels of vegetational cover. The differential reaction patterns indicate changes in the community structure and thus the ecological function of the biotic soil communities, which in turn can affect the entire ecosystem. A “homogenisation” of the species composition of different microhabitats and thus a reduction in the total diversity of specific localities was sporadically observed. Larger penguin and seal colonies were present in many of the studied areas. Therefore, all identified anthropogenic impacts lay above and beyond those caused by wild animals.

Specific recommendations for a better protection of the sensitive Antarctic ecosystems from human impacts could be derived from the study results. These concern, firstly, biosecurity measures against the transfer of non-native soil organisms. Improvements in the information and education of Antarctic visitors regarding these measures and the control of their proper use are necessary. This includes an intensification of their use between Subantarctic and Aantarctic areas as well as after visiting high-risk localities such as Deception Island and Neko Harbour. Furthermore, special microhabitats must be more strongly protected, e.g., by an expansion of areas closed to visitors that contain initial or sporadic vegetation or around meltwater streams. So that a further dispersal of species not native to Antarctica can be prevented and anthropogenic impacts on terrestrial Antarctic habitats be limited in the future, a fundamental limitation of areas that may be visited by tourists is required. For this, a “positive list” is recommended, beyond which touristic visits or expeditions should be prohibited. Furthermore, the establishment of an international, long-term soil-biological monitoring program is recommended, through which a better understanding of long-term human impacts in areas with frequent tourism as well as a control of the success and improvements of the biosecurity measures can be achieved.

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References

Adams, B.J., Bardgett, R.D., Ayres, E., Wall, D. H., Aislabie, J., Bamforth, S., Bargagli, R., Cary, C., Cavacini, P., Connell, L., Convey, P., Fell, J.W., Frati, F, Hogg, I.D., Newsham, K.K., O'Donnell, A., Russell, N., Seppelt, R.D., Stevens, M.I. (2006): Diversity and distribution of Victoria Land biota. Soil Biol. Biochem. 38: 3003–3018.

Anderson, M.J., Gorley R.N., Clarke K.R. (2008): PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E Ltd, Plymouth, UK.

Andrássy, I. (1998): Nematodes in the sixth continent. Journal of Nematode Morphology and Systematics 1: 107–186.

Andrássy, I. (2008): *Eudorylaimus* species (Nematoda: *Dorylaimida*) of continental Antarctica. J. Nematode Morphol. Syst. 11: 49–66.

André, H.M., Noti, M.I., Lebrun, P. (1994): The soil fauna: the other last biotic frontier. Biodiversity and Conservation 3: 45–56.

Aptroot, A., van der Knaap, W.O. (1993): The lichen flora of Deception Island, South Shetland Islands. Nova Hedwigia 56: 183–192.

Argentina (2006): Tourism development in the Antarctic Peninsula: a regional approach. Antarctic Treaty Consultative Meeting XXIX, IP 031: 6 pp.

ATCM (Antarctic Treaty Consultative Meeting) (2005): Site Guidelines for Land-Based Tourist-Visited Sites. Antarctic Treaty Consultative Meeting XXVIII, WP 31: 7 pp.

ATCM (Antarctic Treaty Consultative Meeting) (2006): Report of the CEP Intersessional Contact Group on Site Guidelines for Visitors to Antarctica. Antarctic Treaty Consultative Meeting XXIX, WP 1: 18 pp.

ATCP (Antarctic Treaty Consultative Parties) (1966): Site guidelines for visitors (Resolution 27). Santiago, Chile. www.ats.aq.

ATME (Antarctic Treaty Meeting of Experts) (2004): Chairman's Report from Antarctic Treaty Meeting of Experts on Tourism and Non-governmental Activities in Antarctica: 25 pp.

Ayres, E., Wall, D.H., Adams, B.J., Barrett, J.E., Virginia, R.A. (2007): Unique Similarity of Faunal Communities across Aquatic-Terrestrial Interfaces in a Polar Desert Ecosystem. Ecosystems 10: 523–535.

Ayres, E., Nkem, J.N., Wall, D.H., Adams, B.J., Barrett, J.E., Broos, E.J., Parsons, A.N., Powers, L.E., Simmons, B.L., Virgina, R.A. (2008): Effects of human trampling on populations of soil fauna in the McMurdo Dry Valleys, Antarctica. Conserv. Biol. 22: 1544–1551.

Babenko, A.B. (2003): Landscape Chorology of Springtails in the Taimyr Peninsula: 1. Biotopic Distribution of Species. Entomological Review 83 (8): 1003–1021.

Babenko, A.B. (2010): The Springtail (Hexapoda, Collembola) Fauna of Wrangel Island. Entomological Review 90 (5): 571–584.

Babenko, A.B., Bulavintsev, V. (1993): Fauna and populations of Collembola on the Novaya Zemlya Archipelago. Russian Entomological Journal 2 (3-4): 3–19.

Babenko, A.B., Fjellberg, A. (2006): Collembola septentrionale: A Catalogue of Springtails of the Arctic Region. KMK Press, Moscow.

Baermann, G. (1917): Eine einfache Methode zur Auffindung von Ankylstommum (Nematoda) Larven in Erdproben. Geneesk. Tidjschr. Nederl. Indie 57: 131–137.

Baker, P.E., McReath, I., Roobol, J.J. Davies, T.G. (1975): The geology of the South Shetland Islands:V. volcanic evolution of Deception Island. British Antarctic Survey Scientific Reports No. 78. Cambridge, United Kingdom.

Bardgett, R. (2005): The Biology of Soils. Oxford Univ. Press, Oxford, New York: 242 pp.

Bargagli, R. (2005): Antarctic ecosystems: environmental contamination, climate change, and human impact. Springer, Berlin Heidelberg New York: 395 pp.

Barrett, J.E., Virginia, R.A., Hopkins, D.W., Aislabie, J., Bargagli, R., Bockheim, J.G., Campbell, I.B., Lyons, W.B., Moorhead, D.L., Nkem, J.N., Sletten, R.S., Steltzer, H., Wall, D.H., Wallenstein, M.D. (2006): Terrrestrail ecosystem processes of Victoria Land, Antarctica. Soil Biology & Biochemistry 38: 3019–3034.

Barrett, J.E., Virginia, R.A., Wall, D.H., Adams, B.J. (2008): Decline in a dominant invertebrate species contributes to altered carbon cycling in a low-diversity soil ecosystem. Global Change Biol. 14: 1734–1744.

Bednarek-Ochyra, H., Váňa, J., Ochyra, R., Lewis Smith, R.I. (2000): The Liverwort Flora of Antarctica. Polish Academy of Sciences, Institute of Botany, Cracow: 236 pp.

Bellinger, P.F., Christiansen, K.A. Janssens, F. (2011): Checklist of the Collembola of the World. http://www.collembola.org, accessed July 2011.

Benoit, J.B., Elnitsky, M.A., Schulte, G.G., Lee, R.E., Jr., Denlinger, D.L. (2009): Antarctic Collembolans use chemical signals to promote aggregation and egg laying. Journal of Insect Behavior 22: 121–133.

Bergstrom, D.M., Chown, S.L. (1999): Life at the front: history, ecology and change on southern ocean islands. Trends in Ecology and Evolution 14: 472–476.

Beyer, L., Bölter, M. (2002): Geoecology of Antarctic Ice-Free Coastal Landscapes. Springer, Berlin Heidelberg New York: 427 pp.

Block, W. (1980): *Nanorchestes antarcticus* Strandtmann (Prostigmata) from antarctic ice. Acarologia 21: 173–176.

Block, W. (1982): The Signy Island terrestrial reference sites XIV. Population studies on Collembola. Br. Antarct. Surv. Bull. 55: 33–49.

Block, W. (1984a): An insect introduction to the maritime Antarctic. Biological Journal of the Linnean Society 23: 33–39.

Block, W. (1984b): Terrestrial microbiology, invertebrates and ecosystems. In: Laws, R.M. (ed.): Antarctic Ecology. Academic Press, London: 163–236.

Block, W., Convey, P. (2001): Seasonal and long-term variation in body water-content of an Antarctic springtail - a response to climatic change. Polar Biology 24: 764–770.

Block, W., Harrisson, P.M. (1995): Collembolan water relations and environmental-change in the maritime Antarctic. Global Change Biology 1: 347–359.

Block, W., Starý, J. (1996): Oribatid mites (Acari: Oribatida) of the maritime Antarctic and Antarctic peninsula. Journal of Natural History 30: 1059–1067.

Block, W., Worland, M.R. (2001): Experimental Studies of Ice Nucleation in an Antarctic Springtail. Cryobiology 42: 170–181.

Bokhorst, S., Ronfort, C., Houskes, A., Convey, P., Aerts, R. (2007): Food choice of Antarctic soil arthropods clarified by stable isotope signatures. Polar Biology 30: 983–990.

Booth, R.G. (1984): A new species of mite in the genus *Nanorchestes* (Acari, Prostigmata) from Signy Island, South Orkney Islands. Br. Antarct. Surv. Bull. 63: 111–116.

Booth, R.G., Usher, M.B. (1984): Arthropod communities in a maritime Antarctic moss-turf habitat: effects of the physical and chemical environment. Journal of Animal Ecology 53: 879–893.

Booth, R. G., Usher, M.B. (1986): Arthropod community in a maritime antarctic Moss-turf habitat: Life history strategies of the prostigmatid mites. Pedobiologia 29: 209–218.

Booth, R.G., Edwards, M., Usher, M.B. (1985): Mites of the genus *Eupodes* (Acari, Prostigmata) from maritime Antarctica: a biometrical and taxonomic study. Journal of Zoology, London 207: 381–406.

Boström, S. (1995): Populations of *Plectus acuminatus* Bastian, 1865 and *Panagrolaimus magnivulvatus* n. sp. (Nematoda) from nunataks in Dronning Maud Land, East Antarctica. Fundam. appl. Nematol. 18: 25–34.

Boström, S. (1996): One new and two known nematode species from the Subantarctic Islands South Georgia and East Falkland Island. Fundam. Appl. Nematol. 19: 151–158.

Bowra, G.T., Holdgate, M.W., Tilbrook, P.J. (1966): Biological Investigations in Tottanfjella and Central Heimefrontfjella. Antarct.Surv.Bull. 9: 63–70.

Broady, P.A. (1979): Feeding studies on the collembolan *Cryptopygus antarcticus* Willem at Signy Island, South Orkney Islands. British Antarctic Survey Bulletin 48: 37–46.

Bulavintsev, V.I. (1990): Complexes of microarthropods of the Fildes Peninsula, Ardley and Nelson Island found during ecological monitoring in Antarctica. Sovetskaia Antarktischeskaia Ekspeditsiia, Informatsionny Biulleten 113: 101–108 [In Russian].

Burn, A.J. (1982): A cautionary tale - two recent introductions to the maritime Antarctic. Comité National Français des Recherches Antarctiques 51: 521.

Burn, A.J. (1984): Life cycle strategies in two Antarctic Collembola. Oecologia. 64: 223–229.

Burn, A.J., Lister, A. (1988): Activity patterns in an Antarctic arthropod community. Antarct. Surv. Bull. 78: 43–48

Campbell, I.B., Claridge, G.G.C., Balks, M.R. (1998): Short- and long-term impacts of human disturbances on snow-free surfaces in Antarctica. Polar Record 34: 15–24.

Cannone, N., Ellis Evans, J.C., Strachan, R., Guglielmin, M. (2006): Interactions between climate, vegetation and the active layer in soils at two Maritime Antarctic sites. Antarctic Science 18 (3): 323–333.

Carlini, A.R., Coria, N.R., Santos, M.M., Libertelli, M.M., Donini, G. (2007): Breeding success and population trends in Adélie penguins in areas with low and high levels of human disturbance. Polar Biology 30: 917–924.

Carpenter, G.H. (1907): Collembola from the South Orkney Islands. Proceedings of the Royal Society of Edinburgh 1905/6: 473–483.

Carpenter, G.H. (1909): On some subantarctic Collembola. In Chilton, C. (Ed.) Subantarctic Islands of New Zealand, Vol. 1: Philosophical Institute of Canterbury, Wellington: 377–383.

Caruso, T., Bargagli, R. (2007): Assessing abundance and diversity patterns of soil microarthropod assemblages in northern Victoria Land (Antarctica). Polar Biology 30: 895–902.

Caruso, T., Borghini, F., Bucci, C., Colacevich, A., Bargagli, R. (2007): Modelling local-scale determinants and the probability of microarthropod species occurrence in Antarctic soils. Soil Biology & Biochemistry 39: 2949–2956.

Caruso, T., Garlaschelli, D., Bargagli, R., Convey P. (2009): Testing metabolic scaling theory using intraspecific allometries in Antarctic microarthropods. Oikos 119: 935–945.

Caruso, T., Hogg, I.D., Carapelli, A., Frati, F., Bargagli, R. (2009): Large-scale spatial patterns in the distribution of Collembola (Hexapoda) species in Antarctic terrestrial ecosystems. Journal of Biogeography 36: 879–886.

Cepeda-Pizzaro, J.G., Whitford, W.G (1989): Spatial and temporal variability of higher microarthropod taxa along a transect in a northern Chihuahuan Desert watershed. Pedobiologia 33: 101–111.

Cepeda-Pizzaro, J.G., Gutiérrez, J.R., Valderrama, L., Vasquez , H. (1996): Phenology of the edaphic microarthropods in a Chilean coastal desert site and their response to water and nutrient amendments to the soil. Pedobiologia 40: 352–363.

Chahartaghi, M., Maraun, M., Scheu, S., Domes, K. (2009): Resource depletion and colonization: A comparison between parthenogenetic and sexual Collembola species. Pedobiologia, 52: 181–189.

Chen, J., Blume, H.-P. (1997): Impact of human activities on the terrestrial ecosystem of Antarctica: a review. Polarforschung 65: 83–92.

Chipev, N., Peneva, V., Nedelchev, S. (1996): Abundance and distribution of nematode feeding groups in relation to habitat on Livingston Island, South Shetland Islands, Antarctica. Bulgarian Antarctic Research Life Sciences: 82–92.

Chown, S.L., Convey, P. (2007): Spatial and temporal variability across life's hierarchies in the terrestrial Antarctic. Philosophical Transactions of the Royal Society of London B 362: 2307–2331.

Chown, S.L., Huiskes, A.H.L., Gremmen, N.J.M., Lee, J.E., Terauds, A., Crosbie, K., Frenot, Y., Hughes, K.A., Imura, S., Kiefer, K., Lebouvier, M., Raymond, B., Tsujimoto, M., Ware, C., Vijver van de, B., Bergstrom, D.M. (2012): Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. Proceedings of the National Academy of Science of the United States - Early Edition: 1–6.

Chwedorzewska K.J., Korczak, M. (2010): Human impact upon the environment in the vicinity of Arctowski Station, King George Island, Antarctica. Polish Polar Research 31: 45–60.

Clarke, K.R, Warwick, R.M. (2001): Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. 2nd ed., PRIMER-E Ltd, Plymouth, UK.

Cobley, N.D., Shears, J.R. (1999): Breeding performance of gentoo penguins (*Pygoscelis papua*) at a colony exposed to high levels of human disturbance. Polar Biology 21: 355–360.

Convey, P. (1996): The influence of environmental characteristics on the life history attributes of Antarctic terrestrial biota. Biological Reviews 71: 191–225.

Convey, P. (2001): Antarctic Ecosystems. In: Levin, S.A. (ed.) Encyclopedia of biodiversity, Vol. I. Academic Press, San Diego S. 171–184.

Convey, P. (2005): Antarctic Terrestrial Ecosystems: Responses to Environmental Change. Polarforschung 75: 101–111.

Convey, P. (2006): Antarctic climate change and its influences on terrestrial ecosystems. In: Bergstrom D., Convey, P., Huiskes, A. (eds): Trends in Antarctic terrestrial and limnetic ecosystems: Antarctica as a global indicator. Springer, Dordrecht: 253–272.

Convey, P. (2011): Antarctic terrestrial biodiversity in a changing world. Polar Biology 34: 1629–1641.

Convey, P., Lebouvier, M. (2009): Environmental change and human impacts on terrestrial ecosystems of the sub-Antarctic islands between their discovery and the mid-Twentieth Century. Papers and Proceedings of the Royal Society of Tasmania 143: 33–44.

Convey, P., McInnes, S.J. (2005): Exceptional tardigrade-dominated ecosystems in Ellsworth Land, Antarctica. Ecology 86: 519–527.

Convey, P., Quintana, R.D. (1997): The terrestrial arthropod fauna of Cierva Point SSSI, Danco Coast, northern Antarctic Peninsula. Eur. J. Soil Biology 33: 12–29.

Convey, P., Smith, R.I.L. (1997): The terrestrial arthropod fauna and its habitats in northern Marguerite Bay and Alexander Island, maritime Antarctic. Antarctic Science 9: 12–26.

Convey, P., Smith, R.I.L. (2006): Thermal relationships of bryophytes from geothermal habitats in the South Sandwich Islands, maritime Antarctic. Journal of Vegetation Science 17: 529–538.

Convey, P., Stevens, M.I. (2007): Antarctic Biodiversity. Science 317: 1877–1878.

Convey, P., Greenslade, P., Arnold, R.J., Block, W. (1999): Collembola of sub-Antarctic South Georgia. Polar Biology 22: 1–6.

Convey, P., Greenslade, P., Richard, K.J., Block, W. (1996): The terrestrial arthropod fauna of the Byers Peninsula, Livingston Island, South Shetland Islands - Collembola. Polar Biology 16: 257–259.

Convey, P., Greenslade, P., Pugh, P.J.A. (2000a): The terrestrial micro-arthropod fauna of the South Sandwich Islands. Journal of Natural History 34: 597–609.

Convey, P., Smith, R.I.L., Peat, H.J., Pugh, P.J.A. (2000b): The terrestrial biota of Charcot Island, eastern Bellingshausen Sea, Antarctica: an example of extreme isolation. Antarctic Science 12: 406–413.

Convey, P., Smith, R.I.L., Hodgson, D.A., Peat, H.J. (2000c): The flora of the South Sandwich Islands, with particular reference to the influence of geothermal heating. Journal of Biogeography 27: 1279–1295.

Coulson, S.J., Aakra, K., Aarvik, L., Ávila-Jiménez, M.L., Babenko, B., Biersma,E., Boström, S., Brittain, J., Carlsson, A., Christoffersen, K., Convey, P., De Smet, W., Fjellberg, A., Fuereder, L., Gustafsson, D., Gwiazdowizc, D.J., Hågvar, S., Hansen, L.O., Kaczmarek, L., Kuklin, V., Lakka, H-K., Lebedeva, N., Makarova, O., Maraldo, K., Melekhina, E., Ødegaard, F., Pilskog, H.E., Simon, J.C., Solenius, B., Solhøy, T., Søli, G., Stur. E., Tanasevitch, A., Taskaeva, A., Velle, G., Zmudczyńska, K.: The terrestrial and freshwater invertebrate biodiversity of the archipelagoes of the Barents Sea; Svalbard, Franz Josef Land and Novaya Zemlya (in press).

Crafford, J.E. (1986): A case study of an alien invertebrate (*Limnophyes pusillus*, Diptera, Chironomidae) introduced on Marion Island: Selective advantages. South African journal of antarctic research 16: 115–117.

Cross, E.A. (1964): Pyemotidae (Acarina: Tarsonemoidea) from Macquarie Island. Pacific Insects Monograph 7 (Supplement): 642–646.

Cross, E.A. (1970): Acarina: Prostigmata: Tarsonemoidea: Pyemotidae of South Georgia. Pacific Insects Monograph 23: 115–120.

Daleniu, P., Wilson, O. (1958): On the soil fauna of the Antarctic and of the Sub-Antarctic Islands. The Oribatidae (Acari). Arkiv för Zoologi. N.S. 11: 393–425.

Dalenius, P. (1965): The Acarology of the Antarctic Regions. Monographiae Biologicae; Biogeography and Ecology in Antarctica 15: 414–430.

Dallai, R., Malatesta, E., Focardi, S. (1988): On two Antarctic Collembola *Gressittacantha terranova* and *Friesea grisea*. Revue de Écology et de Biology du sol 25(3): 365–372.

Dastych, H. (1984): The Tardigrada from Antarctic with descriptions of several new species. *Acta Zoologica Cracoviensia* 27(19): 377–436.

Dastych, H. (1989): An annotated list of Tardigrada from the Antarctic. Entomologische Mitteilungen aus dem Zoologischen Museum Hamburg 9: 136–137.

Davies, K.F., Greenslade, P., Melbourne, B.A. (1997): The invertebrates of sub-Antarctic Bishop Island. Polar Biology 17: 455–458.

Day, T., Ruhland, C.T., Strauss, S.L., Park, J.-H., Krieg, M.L., Krna, M.A., Bryant, D.M. (2009): Response of plants and the dominant microarthropod, *Cryptopygus antarcticus*, to warming and contrasting precipitation regimes in Antarctic tundra. Global Change Biology 15: 1640–1651.

de Man, J.G. (1904): Nématodes libres. In: Résultats du voyage du S. Y. Belgica en 1897-1898-1899 sous le commendemant de A. de Gerlache de Gomery. Zoologie: 1–55.

de Poorter, M., Gilbert, N., Storey, B., Rogan-Finnemore, M. (2006): Non-native Species in the Antarctic: Workshop Final Report. Gateway Antarctica, University of Canterbury: 40 pp.

de Villiers, M. (2008): Review of recent research into the effects of human disturbance on wildlife in the Antarctic and sub-Antarctic region. In: Human disturbance to wildlife in the broader Antarctic region: a review of findings. Appendix 1. Antarctic Treaty Consultative Meeting XXXI, WP 12: 41 pp.

Deharveng, L. (1981): Collemboles des Iles subantarctiques de l'Océan Indien. Comité National Français des Recherches Antarctiques. 48: 33–109.

Delamare Debouteville, C., Massoud, Z. (1966): Les Collemboles de l'Ile de la Possession et de I'le aux Cochons. CNFR. 15 (7): 65–72.

Deutsches Institut für Normung e. V. (1996): DIN ISO 10694, Berlin

Deutsches Institut für Normung e. V. (2005): DIN ISO 10390, Berlin

Downie, R.H. (2002): An Overview of the Flora and Fauna of Deception Island 24–32. In: Valencia, J., Downie, R. (Eds.) Workshop on a Management Plan for Deception Island. Instituto Antártico Chileno pp. 1–109.

Downie, R.H., Convey, P., McInnes, S.J., Pugh, P.J.A. (2000): The non-marine invertebrate fauna of Deception Island (Maritime Antarctic): a baseline for a comprehensive biodiversity database. Polar Record 36 (199): 297–304.

Dózsa-Farkas, K., Convey, P. (1997): *Christensenia*, a new terrestrial enchytraeid genus from Antarctica. Polar Biology 17: 482–486.

Dunger, W., Schlitt, B. (2011): Synopses on Palaearctic Collembola. vol. 6/1: Tullbergiidae. Soil Organisms 83 (1): 1–168.

Ellis-Evans, J.C., Walton, D. (1990): The process of colonization in Antarctic terrestrial and freshwater ecosystems. Proceedings of the NIPR Symposium on Polar Biology 3: 151–163.

Elmer, M., Hohberg, K., Russell, D., Christian, A., Schulz, H.-J., Wanner, M. (2010): Succession of the soil faunal community during initial ecosystem development. In: Schaaf, W., Biemelt, D., Hüttl, R.F. (Eds.): Initial development of the artificial catchment 'Chicken Creek' - monitoring program and survey 2005-2008. Ecosystem Development 2: 97–118.

Erize, F.J. (1987): The impact of tourism on the Antarctic environment. Environmental International 13: 133–136.

Estrada, E.G., Sánchez, I., Bassols, I. (1988): Acaros del suelo de dos zonas del Valle de Tehuacan, Puebla, Mexico. Folia Entomol. Mex. 76: 225–236.

Fitzsimons, J.M. (1971): On the food habits of certain Antarctic Arthropods from coastal Victoria Land and Adjacent Islands. Pacific Insects Monograph 25: 121-125

Fjellberg, A. (1994): Collembola of Norwegian Arctic Islands. Meddelelser No. 133, Oslo.

Fjellberg, A. (1998): The Collembola of Fennoscandia and Denmark. Part I: Poduromorpha. Fauna Entomologica Scandinavica, 35: 184 pp.

Fjellberg, A. (2007): The Collembola of Fennoscandia and Denmark. Part. II:Entomybryomorpha and Symphypleona . Fauna Entomologica Scandinavia 42: 264 pp.

Fox, A.J., Paul, A., Cooper, R. (1994): Measured properties of the Antarctic ice sheet derived from the SCAR Antarctic digital database. Polar Record 30: 201–206.

Franco, P.J., Edney, E.B., McBrayer, J.F. (1979): The distribution and abundance of soil arthopods in the Northern Mojave desert. J. Arid Environ. 2: 137–149

Fraser, W.R., Patterson-Fraser, D.L. (2009): Distinguishing Human Impacts at Palmer Station, Antarctica. Antarctic Treaty Consultative Meeting XXXII, IP 80: 9 pp.

Frati, F., Fanciulli, P.P., Carapelli, A., Dallai R. (1997): The Collembola of northern Victoria Land (Antarctica): distribution and ecological remarks. Pedobiologia 41: 50–55.

Freckman, D.W., Virginia, R.A. (1997): Low-diversity Antarctic soil nematode communities: distribution and response to disturbance. Ecology 78: 363–369.

Frenot, Y., Chown, S.L., Whinam, J., Selkirk, P., Convey, P., Skotnicki, M., Bergstrom, D.M. (2005): Biological invasions in the Antarctic: extent, impacts and implications. Biological Reviews 80: 45–72.

Gabriel, A.G.A., Chown, S.L., Barendese, J., Marshall, D.J., Mercer, R.D., Pugh, P.J.A., Smith, V.R. (2001): Biological invasions of Southern Ocean islands, the Collembola of Marion Island as a test of generalities. Ecography 24: 421–430.

Giller, P.S. (1996): The diversity of soil communities, the "poor man's tropical forest". Biodiversity and Conservation 5: 135–168.

Goddard, D.G. (1979a): The Signy Island terrestrial reference sites: XI. Population studies on the Acari. Antarct. Surv. Bull. 48: 71–92.

Goddard, D.G. (1979b): Biological observations on the free-living mites of signy island. Antarct. Surv. Bull. 49: 181–205.

Goddard, D.G. (1982): Feeding biology of free-living Acari at Signy Island, South Orkney Islands. British Antarctic Survey Bulletin 51: 290–293.

Greenslade, P. (1992): New records of *Mesaphorura* (Collembola: Onychiuridae, Tullbergiinae) species from Australia, Macquarie Island and the Antarctic. Transactions of the Royal Society of South Australia. 116: 141–143.

Greenslade, P. (1995): Collembola from the Scotia Arc and Antarctic Peninsula including descriptions of two new species and notes on biogeography. Polskie Pismo Entomologiczne, 64: 305–319.

Greenslade, P. (2006): Invertebrates of Macquarie Island. Kingston, TAS: Australian Antarctic Division: 326 pp.

Greenslade, P. (2010): South Shetlands Collembola fauna revisited. Antarctic Science 22: 233–242.

Greenslade, P., Convey, P. (2011): Exotic Collembola on subantarctic islands: pathways, origins and biology. Biological Invasions. DOI 10.1007/s10530-011-0086-8.

Greenslade, P., Convey, P. (2012): Exotic Collembola on subantarctic islands: pathways, origins and biology. Biological Invasions 14: 405–417.

Greenslade, P., Wise, K.A.J. (1984): Additions to the collembolan fauna of the Antarctic. Transactions of the Royal Society of South Australia 108: 203–205.

Gressitt, J.L. (1967): Notes on arthropod populations in the Antarctic Peninsular-South Shetland Islands-South Orkney Islands area. Entomology of Antarctica (Antarctic research series) 10: 373–391

Gressitt, J.l., Weber, N.A. (1959): Bibliographic Introduction to Antarctic-Subantarctic Entomology. Pacific Insects. 1(4): 441–480.

Gressitt, J.L., Yoshimoto, C.M. (1963): Dispersal of animals in the pacific. Pacific Basin Biogeography - a Symposium: 283–292.

Gressitt, J.L., Leech, R.E., Wise, K.A.J. (1963): Entomological investigations in Antarctica. Pacific Insects 5 (1): 287–304

Gressitt, J.L., Maa, T.C., Mackerras, I.M., Nakata, S., Quate, L.W. (1961): Problems in the Zoogeography of Pacific and Antarctic insects. Pacific Insects Monograph 2: 1–94.

Gryziak, G. (2009): Colonization by mites glacier-free areas in King George Island, Antarctica. Pesq. agropec. bras., Brasilia 44: 891–895.

Hack, W.H. (1949): Nota sobre un colémbolo de la Antartida Argentina *Achorutes viaticus* Tullberg. Notas del Museo de la Plata 14: 211–212.

Hallan, J. (2005): Synopsis of the described Actinedida of the world. http://insects.tamu.edu/research/collection/hallan/acari/Actinedida1.htm

Hammer, M. (1958): Investigations on the oribatid fauna of the Andes mountains. I. The Argentine and Bolivia. Biologiske Skrifter udgivet af Det Kongelige Danske Videnskabernes Selskab 10: 1–129.

Hart, I.B. (2006): Whaling in the Falkland Islands Dependencies 1904–1931. A history of shore and bay-based whaling in the Antarctic. Pequena, Newton St. Margarets, United Kingdom.

Hässel de Menendez, G.G. (1977): Liverworts new to South Georgia. – British Antarctic Survey Bulletin 46: 99–108.

Hawes, T.D., Worland, M.R., Convey, P., Bale, J.S. (2007): Aerial dispersal of springtails on the Antarctic Peninsula: implications for local distribution and demography. Antarctic Science 19: 3–10.

Hawes, T.C., Bale, J.S., Worland, M.R., Convey, P. (2008): Trade-offs between microhabitat selection and physiological plasticity in the Antarctic springtail, *Cryptopygus antarcticus* (Willem). Polar Biology 31: 681–689.

Hayward, S.A.L., Worland, R.M., Convey, P., Jeff, S. (2004): Habitat moisture availability and the local distribution of the Antarctic Collembola *Cryptopygus antarcticus* and *Friesea grisea*. Soil Biology and Biochemistry 30. 927–934.

Hemmings, A.D., Roura, R. (2003): A square peg in a round hole: fitting impact assessment under the Antarctic Environmental Protocol to Antarctic tourism. Impact Assessment and Project Appraisal 21: 13–24.

Henssen, A., Renner, B. (1981): Studies in the lichen genus Psoroma I. *Psoroma tenue* and *Psoroma cinnamomeum*. – Mycotaxon 13: 433–449.

Higashi, S., Sugawara, H. (1992): Springtails in the Antarctica. Insectarium 29: 52–58.

Hiruta, S., Ohyama, Y (1995): A preliminary report on terrestrial invertebrates in the Asuka Station area, Antarctica. Proc. NIPR Symp. Polar Biol. 8: 188–193.

Hofman, R., Jatko, J. (2000): Assessment of the possible cumulative environmental impacts of commercial ship-based tourism in the Antarctic Peninsula Area. National Science Foundation, Washington: 94 pp.

Hogg, I.D., Stevens, M.I. (2002): Soil Fauna of Antarctic Coastal Landscapes. In: Beyer, L. & M. Bölter (eds): Geoecology of Antarctic Ice-Free Coastal Landscapes. Springer, Berlin Heidelberg New York: 265–280.

Hogg, I.D., Cary, S.C., Convey, P., Newsham, K.K., O'Donnell, A.G., Adams, B.J., Aislabie, J., Frati, F., Stevens, M.I., Wall, D.H. (2006): Biotic interactions in Antarctic terrestrial ecosystems: Are they a factor? Soil Biology & Biochemistry 38: 3035–3040.

Holmes, N.D., Giese, M.A., Achurch, H., Robinson, S., Kriwoken, L. (2006): Behaviour and breeding success of gentoo penguins *Pygoscelis papua* in areas of low and high human activity. Polar Biology 29: 399–412.

Holovachov, O., Boström, S. (2006): Description of *Acrobeloides arctowskii* sp. N. (Rhabditida: Cephalobidae) from King Georg Island, Antarctica. Russian J. Nematol. 14: 51–56.

Hughes K.A., Convey P. (2010): The protection of Antarctic terrestrial ecosystems from inter- and intra-continental transfer of non-indigenous species by human activities: A review of current systems and practices. Global Environmental Change 20: 96–112.

Hughes, K.A. and Convey, P. (2012): Determining the native / non-native status of newly discovered terrestrial and freshwater species in Antarctica - Current knowledge, methodology and management action. Journal of Environmental Management 93: 52–66.

Hughes, K.A., Convey, P., Maslen, N.R., and Smith, R.I.L. (2010): Accidental transfer of non-native soil organisms into Antarctica on construction vehicles. Biological Invasions 12: 875–891.

Hughes, K.A., Lee, J.E., Tsujimoto, M., Imura, S., Bergstrom, D.M., Ware, C., Lebouvier, M., Huiskes, A.H.L., Gremmen, N.J.M., Frenot, Y., Bridge, P.D., Chown, S.L. (2011): Food for thought: Risks of non-native species transfer to the Antarctic region with fresh produce. Biological Conservation 144: 1682–1689.

Hunter, P.E. (1967): Rhodacaridae and Parasitidae mites (Acarina: Mesostigmata) collected by the British Antarctic Survey, 1961-1964. British Antarctic Survey Bulletin 13: 31–39.

IAATO (2001): Issues relating to cumulative environmental impacts of tourist activities. Antarctic Treaty Consultative Meeting XXIV IP 52: 17 pp.

IAATO (2010): Non-IAATO Tourism and Visitation in Antarctica, Antarctic Treaty Consultative Meeting XXXIII, IP 75: 4 pp.

IAATO (2011): IAATO Overview of Antarctic Tourism: 2010-11 Season and Preliminary Estimates for the 2011-12 Season. Antarctic Treaty Consultative Meeting XXXIV, IP 106: 19 pp.

Janetschek, H. (1962): Über die wirbellose Landfauna des Rossmeergebietes (Antarctika). Anzeiger für Schädlingskunde 31: 8–12.

Janetschek, H. (1967): Arthropod ecology of South Victoria Land Antarctica. Entomology of Antarctica. Antarctic research series 10: 205–293.

Jennings, P.G. (1976): Tardigrada from the Antarctic Peninsula and Scotia Ridge region. British Antarctic Survey Bulletin 44: 77–95.

Jennings, P.G. (1979): Signy Island reference sites: X. Population dynamics of Tardigrada and Rotifera. British Antarctic Survey Bulletin 47: 89–105.

Judson, M. (1995): Identity of the Antarctic mite *Gainia nivalis* Trouessart (Acari, Nanorchestidae). Bull. Mus. Natl. Hist. nat. Paris 17: 83–8.

Karg, W., Schorlemmer, A. (2009): New insights into predatory mites (Acarina, Gamasina) from tropical rain forests with special reference to distribution and taxonomy. Zoosystematics and Evolution 85: 57–91.

Kazmierski, A. (1998): Tydeinae of the world: generic relationships, new and redescribed taxa and keys to all species. A revision of the subfamilies Pretydeinae and Tydeinae (Acari: Actinedida: Tydaidae) – part IV. Acta Zoologica Cracoviensia 41: 283–455.

Kennicutt, M.C., Klein, A., Montagna, P., Sweet, S., Wade, T., Palmer, T., Sericano, J., Denoux, G. (2010): Temporal and spatial patterns of anthropogenic disturbance at McMurdo Station, Antarctica. Environmental Research Letters 5: 10 pp.

Kethley, J. (1990): Acarina: Prostigmata (Actinedida). In: Dindal, D.L. (ed.): Soil Biology Guide. Wiley, New York: pp. 667–756.

Kethley, J., Welbourne, W.C. (2010): Artificial key to the families of soil Prostigmata. Unveröff. Manuscript, OSU Acarological Sommer Program: Soil Acarology: 368 pp.

Knox, G.A. (2006): The biology of the Southern Ocean, 2nd ed. CRC Press, Boca Raton: 640 pp.

Kranzt, G.W., Walter, D.E. (2009): A Manual of Acarology. 3rd Edn. Texas Tech Univ. Press, Lubbock: 807 pp.

Kriwoken, L.K., Rootes, D. (2000): Tourism on ice: environmental impact assessment of Antarctic tourism. Impact Assessment and Project Appraisal 18: 138–150.

Lister, A., Block, W., Usher, M.B. (1988): Arthropod predation in an Antarctic terrestrial community. Journal of Animal Ecology 57: 957–971.

Longton, R.E. (1988): The biology of Polar bryophytes and lichens. – Cambridge University Press, Cambridge/New York/New Rochelle/Melbourne/Sydney.

Lynch, H.J., Crosbie, K., Fagan, W.F., Naveen, R. (2009): Spatial patterns of tour ship traffic in the Antarctic Peninsula region. Antarctic Science 22:123–130.

Macfadyen, A. (1961): Improved funnel-type extractors for soil arthropods. J. Anim. Ecol. 30: 171-184.

Maher, H. (2005): Managing Tourism in Antarctica – A Framework for the Future. Master Thesis, Univ. Otago, New Zealand: 154 pp.

Mahunka, S. (1967): *Pygmephorus antarcticus* n.sp., eine neue Milbenart von der Antarktis (Acari: Tarsonemini). Opusc. Zool. Budapest 7: 227–228.

Marshall, D.J., Pugh, P.J.A. (1996): Origin of the inland Acari of Continental Antarctica, with particular reference to Dronning Maud Land. Zoological Journal of the Linnean Society 118: 101–118.

Marshall, D.J, Gremmen, N.J.M., Coetzee, L., OConnor, B.M., Pugh, P.J.A., Theron, P.D., Ueckermann, E.A. (1999): New records of Acari from the sub-Antarctic Prince Edward Island. Polar Biol. 21: 84–89.

Maslen, N.R. (1979a): Six new nematode species from the maritime Antarctic. Nematologica 25: 288–308.

Maslen, N.R. (1979b): Additions to the nematode fauna of the Antarctic Region with keys to taxa. Br. Antarct. Surv. Bull. 49: 207–229.

Maslen, N.R., Convey, P. (2006): Nematode diversity and distribution in the southern maritime Antarctic – clues to history? Soil Biol. Biochem. 38: 3141–3151.

Massoud, Z., Rapoport, E.H. (1968): Collemboles Isotomides d'Amerique du sud et de l'Antarctique. Biol. Amer. Aus. CNRS France 4: 307–337.

McDaniel, B., Theron, P.D. (1979): A new distribution record and a morphological variant of *Terpnacarus glebulentus* Theron (Acari: Terpnacaridae). Entomological News 90: 143–144.

McInnes, S.J., Pugh, P.J.A. (1998): Biogeography of limno-terrestrial Tardigrada, with particular reference to the Antarctic fauna. Journal of Biogeography 25: 31–36.

McInnes, S.J., Pugh, P.J.A. (1999): Zonation in Antarctic lake–dwelling benthic meiofauna, with emphasis on the Tardigrada. Zoologischer Anzeiger 238: 283–288.

McNeill, M., Phillips, C., Young, S., Shah, F., Aalders, L., Bell, N, Gerard, E., Littlejohn, R. (2011): Transportation of nonindigenous species via soil on international aircraft passengers' footwear. Biological Invasions 13: 2799–2815.

Mertens, J., Coessens, R., Blanquaert, J.P. (1983): Reproduction and development of Hypogastrura viatica (Collembola) in relation to temperature and submerged conditions. Revue d'Ecologie et Biologie du Sol 20: 567–577.

Michael, A.D. (1895): Ueber die auf Süd-Georgien von der deutschen Station 1882-1883 gesammelten Oribatiden. Jahrbuch der Hamburgischen Wissenschaftlichen Anstalten 12: 69–72.

Micol, T., Jouventin, P. (2001): Long-term population trends in seven Antarctic seabirds at ointe Geologie (Terre Adélie); human impact compared with environmental change. Polar Biology 24: 175–185.

Murray, C., Jabour, J. (2004): Independent expeditions and Antarctic tourism policy. Polar Record 40: 309–317.

Murray, J. (1906): Scottish National Antarctic Expedition: Tardigrada of the South Orkneys. *Transactions of the Royal Society of Edinburgh* 45: 323-338.

Naveen, R., Forrest, S.C., Dagit, R. G., Blight, L.K., Trivelpiece, W.Z., Trivelpiece, S.G. (2001): Zodiac landings by tourist ships in the Antarctic Peninsula region, 1989-99. Polar Record 37: 121–132.

Naveen, R., Lynch, H. (2011): Antarctic Peninsula Compendium, 3rd Edition. Antarctic Site Inventory - Report to the United States Environmental Protection Agency. Oceanites Inc., Chevy Chase, Maryland: 358 pp.

Nedelchev, S., Peneva, V. (2000): Description of three new species of the genus *Mesodorylaimus* Andrássy, 1959 (Nematoda: Dorylaimidae) from Livington Island, Antarctica, with notes on *M. imperator* Loof, 1975. Russian J. Nematol. 8: 161–172.

Nedelchev, S., Peneva, V. (2007): Description of the male of *Mesodorylaimus chipevi* Nedelchev & Peneva, 2000 (Nematoda: Dorylaimidae) from Livingston Island, Antarctica. Russian J. Nematol. 15: 75–78.

Nkem, J.N., Wall, D.H., Virginia, R. A., Barrett, J.E., Broos, E.J., Porazinska, D.L.,Adams, B.J. (2006): Wind dispersal of soil invertebrates in the McMurdo Dry Valleys, Antarctica. Polar Biol. 29: 346–352.

Noble, J.C., Whitford, W.G., Kaliszweski, M. (1996): Soil and litter microarthropod populations from two contrasting ecosystems in semi-arid eastern Australia. J. Arid Environ. 32: 329–346.

Norton, R.A., Behan-Pelletier, V.M. (2009): Suborder Oribatida. In: Krantz, G.W., Walter, D.E. (eds) A Manual of Acarology, 3rd ed. Texas Tech University Press, Lubbock, TX, USA: pp. 430–564.

Ochyra, R., Smith, R.I.L., Bednarek-Ochyra, H. (2008): The illustrated Moss Flora of Antarctica. – Cambridge University Press, Cambridge.

Ohyama, Y. (1978): Population density of free-living mites in the ice-free areas around Syowa Station, East Antarctica. National Institute of Polar Research 60: 47–56.

Ohyama, Y., Hiruta, S. (1995): The terrestrial arthropods of Sør Rondane in eastern Dronning Maud Land, Antarctica, with biogeographical notes. Polar Biology 15: 341–347.

Ohyama, Y., Shimada, K. (1998): Ecological survey of Collembolans in King George Island, maritime Antarctic. Edaphologia, 60: 23–35.

Olech, M. (2004): Lichens of King George Island, Antarctica. – Institute of Botany of the Jagiellonian University, Kraków.

Øvstedal, D.O., Smith, R.I.L (2001): Lichens of Antarctica and South Georgia. A guide to their Identification and Ecology. – Cambridge University Press, Cambridge.

Peat, H.J., Clarke, A., Convey, P. (2007): Diversity and biogeography of the Antarctic flora. Journal of Biogeography 34: 132–146.

Peneva, V., Chipev, N., Nedelchev, S. (1996): *Prionchulus punctatus* Cobb, 1917, a new record from Patagonia (Chile), with notes on other mononchids from Antarctic Regions.

Peter, H.-U., Buesser, C., Mustafa, O., Pfeiffer, S. (2008): Risk assessment for the Fildes Peninsula and Ardley Island, and development of management plans for their designation as Specially Protected or Specially Managed Areas. UBA Texte 20/08, German Environmental Agency, Dessau. 478 pp.

Peter, H.-U, Braun, C., Janowski, S., Nordt, A., Nordt, A., Stelter, M. (2013): The current environmental situation and proposals for the management of the Fildes Peninsula region. UBA Texte 03/2013, German Environmental Agency, Dessau. 127 pp. + Appendeces

Petersen, H., Luxton, M. (1982): A comparative analysis of soil fauna populations and their role in decomposition processes. Oikos 39 (3): 284–422.

Pfeiffer, S., Peter, H.-U. (2004): Ecological studies toward the management of an Antarctic tourist landing site (Penguin Island, South Shetland Islands). Polar Record 40: 345–353.

Poland, J.S., Riddle, M.J., Zeeb, B.A. (2003): Contaminants in the Arctic and the Antarctic: a comparison of sources, impacts, and remediation options. Polar Record 39: 369–383.

Pomorski, R.J. (1998): Onychiurinae of Poland. Genus (Suppl.): 201 pp.

Pomorski, R.J., Skarzynski, D. (1995): Springtails (Collembola) collected in Chupa Inlet region (N Karelia, Russia). Acta Universitatis Wratislaviensis 1744 (Prace Zoolgiczne 29): 47–57.

Porazinska, D.L, Wall, D.H., Virginia, R.A. (2002): Invertebrates in ornithogenic soils on Ross Island, Antarctica. Polar Biol. 25: 569–574.

Potapov, M. (1991): *Antarctophorus* - a new genus of Isotomidae (Collembola) from Antarctica. Revue Ecologie et Biologie du Sol 28: 491–495.

Potapov, M. (2001). Synopses on Palaearctic Collembola. vol. 3: Isotomidae. Abhandlungen und Berichte des Naturkundemuseums Görlitz. 73: 1–603.

Pugh P.J.A. (1993): A synonymic catalogue of the Acari from Antarctica, the sub-Antarctic Islands and the Southern Ocean. Journal of Natural History 27: 323–421.

Pugh P.J.A. (1994): Non-indigenous Acari of Antarctica and the sub-Antarctic islands. Zoological Journal of the Linnean Society 110: 207–217.

Pugh, P.J.A. (1997): Acarine colonisation of Antarctica and the islands of the Southern Ocean: the role of zoohoria. Polar Record 33. 113–122.

Pugh, P.J.A., Dartnall, H.J.G. (1994): The Acari of fresh and brackish-water habitats in the Antarctic and sub-Antarctic regions. Polar Biology 14: 401–404.

Richard, K.J., Convey, P., Block, W. (1994): The terrestrial arthropod fauna of the Byers Peninsula, Livingston Island, South Shetland Islands. Polar Biology 14: 371–379.

Richters, F. (1904): Vorläufiger Bericht über die antarktische Moosfauna. Verhandlungen der Deutschen Zoologischen Gesellschaft. 24: 236-239.

Roura, R. (2009): Making Tangible Progress on a Strategic Vision for Antarctic Tourism. ATCE IV, IP 10, 4 pp.

Roura, R., Hemmings, A.D., Barnes, J. (2008): A decade of Antarctic tourism: Status, change, and actions needed. Antarctic Treaty Consultative Meeting XXXI, IP41: 15 pp.

Rounsevell, D. (1978): Populations of introduced arthropods at Australien Antarctic stations. Pacific Insects 18: 199–202.

Rounsevell, D.E., Greenslade, P. (1988): Cuticle structure and habitat in the Nanorchestidae (Acari: Prostigmata). Hydrobiologia 165: 209–212.

Russell, D.J., Alberti, G. (2009): Actinedid mite community diversity in a succession gradient in continental sand-dune habitats of central Europe. In: Sabelis, M.W., Bruin, J. (eds.) Trends in Acarology. Springer, Dordrecht pp. 135–142.

Ryan P.G., Watkins, B.P., Smith, R.I.L., Dastych, H., Eicker, A., Foissner, W., Heatwole, H., Miller, W.R., Thompson, G. (1989): Biological survey of Robertskollen western Dronning Maud Land Antarctica area description and preliminary species lists. South African Journal of Antarctic Research 19: 10–20.

Ryss A., Boström, S., Sohlenius, B. (2005): Tylenchid nematodes found on the nunatak Basen, East Antarctica. Annales Zoologici 55(3): 315–324.

Salmon, J. (1949): The Collembola of the United States Antarctic Service Expedition 1939–41. Royal Entomological Society of London, 18: 161–162.

Salmon, J.T. (1962): A new species and redescriptions of Collembola from Antarctica. Pacific Insects, 4: 87–889.

Sanchez-Rocha, I., Palacios-Vargas, J. G. (1996): Edaphic mites (Acari: Prostigmata) from Queretaro State, Mexico (abstract). XII Intl Colloq Soil Zool., Dublin: Pg. 92

Sanyal, A.K. (2004): Diversity of Invertebrate Fauna of Schirmacher Oasis, East Antarctica. Zoological Survey of India 17: 173–187.

Sanyal, A.K., Gupta, S.K. (2005): Acari from Antarctic continent: one new species of *Proctolaelaps* (Mesostigmata: Ascidae) and some new reports (Prostigmata, Astigmata, Mesostigmata, Cryptostimata). Proceedings of the Zoological Society (Calcutta) 58: 79–84.

Sanyal, A.K., Hazra, A.K. (2008): A review on studies on Collembola (Insecta) and mite (Acari) in Schirmacher Oasis, East Antarctica. Bionotes 10: 118–120.

Schaeffer, C. (1891): Die Collembola von Süd-Georgien nach der Ausbeute der deutschen Station von 1882/1883. Jahrbuch der Hamburgischen Wissenschaftlichen Anstalten 9: 193–201.

Schatz, H., Dusbábek, F., Bukva, V. (1991): Arrival and establishment of acari on oceanic islands. Modern Acarology 2 proceedings of the 8 International Congress of Acarology held in Ceske Budejovice, Czechoslovakia, 6-11 August 1990: 613–618.

Schulte, G.G., Elnitsky, M.A., Benoit, J.B., Denlinger, D.L., Lee, R.E. (2008): Extremely large aggregations of collembolan eggs on Humble Island, Antarctica: a response to early seasonal warming? Polar Biology 31: 889–892.

Simmons, B.L., Wall, D.H., Adams, B.J., Ayres, E., Barrett, J.E., Virginia, R.A. (2009). Terrestrial mesofauna in above- and below-ground habitats: Taylor Valley, Antarctica. Polar Biol. 32: 1549–1558.

Sinclair, B.J. (2001): On the distribution of terrestrial invertebrates at Cape Bird, Ross Island, Antarctica. Polar Biology 24: 394–400.

Sinclair, B.J., Stevens, M.I. (2006): Terrestrial microarthropods of Victoria Land and Queen Maud Mountains, Antarctica: Implications of climate change. Soil Biology & Biochemistry 38: 3158–3170.

Sinclair, B.J., Scott, M.B., Klok, C. J., Terblanche, J.S., Marshall D.J., Reyers, B., Chown, S.L. (2006): Determinants of terrestrial arthropod community composition at Cape Hallett, Antarctica. Antarctic Science 18: 303–312.

Slabber, S., Chown, S.L. (2002): The first record of a terrestrial crustacean, *Porcellio scaber* Isopoda, Porcellionidae), from sub Antarctic Marion Island. Polar Biol. 25: 855–858.

Smellie, J.L., López-Martínez, J., Headland, R.K., Hernández-Cifuentes, F., Maestro, A., Millar, I.L., Rey, J., Serrano, E., Somoza, L. Thomson, J.W. (2002): Geology and geomorphology of Deception Island. BAS GEOMAP Series, Sheets 6-A and 6-B,1:25000. Cambridge, British Antarctic Survey: 78 pp.

Smith, R.I.L. (2005a): The bryophyte flora of geothermal habitats on Deception Island, Antarctica. Journal of the Hattori Botanical Laboratory 97: 233–248.

Smith, R.I.L. (2005b): The thermophilic bryoflora of Deception Island: unique plant communities as a criterion for designating an Antarctic Specially Protected Area. Antarctic Science 17: 17–27.

Smith, T., Wharton, D.A., Marshall, C.J. (2008): Cold tolerance of an Antarctic nematode that survives intracellular freezing: comparisons with other nematode species. J. Comp. Physiol. B 178: 93–100.

Sohlenius, B., Bostrom, S., Hirschfelder, A. (1995): Nematodes, Rotifers and Tardigrades from nunataks in Dronning-Maud-Land, East Antarctica. Polar Biology 15: 51–56.

Sohlenius, B., Boström, S., Jönsson, K.I. (2004): Occurrence of nematodes, tardigrades and rotifers on ice-free areas in East Antarctica. Pedobiologia 48: 395–408.

Sohlenius, B., Bostrom, S. (2008): Species diversity and random distribution of microfauna in extremely isolated habitable patches on Antarctic nunataks. Polar Biology 31: 817–825.

Sømme, L. (1986): New records of terrestrial arthropods from Dronning Maud Land Antarctica. Polar Research 4: 225–230.

Sømme, L., Stromme, A., Zachariassen, K. (1993): Notes on the ecology and physiology of the Antarctic oribatid mite *Maudheimia wilsoni*. Polar Research 12: 21–25.

Spaull, V.W. (1973): Distribution of soil nematodes in the maritime Antarctic. British Antarctic Survey Bulletin 37: 1–6.

Starý, J., Block, W., Greenslade, P. (1997): Oribatid mites (Acari: Oribatida) of sub-Antatctic Heard Island. Journal of Natural History 31: 545–553.

Starý, J., Block, W. (1998): Distribution and biogeography of oribatid mites (Acari: Oribatida) in Antarctica, the sub-Antarctic islands and nearby land areas. Journal of Natural History 32: 861–894.

Steinberger, Y., Aldon, E.F., Whitford, W.G. (1990): Soil microarthropod fauna of four habitats of the Rio Puerco watershed, New Mexico. Southwest Nat. 35: 279–284.

Stevens, M.I., Hogg, I.D. (2002): Expanded distributional records of Collembola and Acari in southern Victoria Land, Antarctica. Pedobiologia 46: 485–495.

Stevens M.I., Hogg I.D. (2006a): Contrasting levels of mitochondrial DNA variability between mites (Penthalodidae) and springtails (Hypogastruridae) from the Trans-Antarctic Mountains suggest long-term effects of glaciation and life history on substitution rates, and speciation processes. Soil Biology & Biochemistry 38: 3171–3180.

Stevens, M.I., Hogg, I.D. (2006b): The molecular ecology of Antarctic terrestrial and limnetic invertebrates. In Bergstrom, D., Convey, P., Huiskes, A., eds. Trends in Antarctic terrestrial and limnetic ecosystems. Dordrecht: Springer: 1–11.

Stevens, M., Greenslade, P., Hogg, I.D., Sunnucks, P. (2005): Southern Hemisphere springtails: could they have survived glaciation of Antarctica? Molecular Biology and Evolution 23: 874–882.

Strandtmann, R.W. (1967): Terrestrial Prostigmata (Trombidiform mites). Antarctic Research Series 10: 51–80.

Strandtmann, R. W. (1968): A comparison of the prostigmatic eupodiforme mites of the north and south polar regions. Proceedings 13th International Congress of Entomology, Moscow 3: 354–354.

Strandtmann, R.W., Tilbrook, P.J. (1968): Some Prostigmata (Acari) from Signy Island, South Orkney Islands, and Deception Island, South Shetland Islands. Antarct. Surv. Bull. 17: 51–57.

Strandtmann, R.W., Pittard, D., Schaefer, P. (1967): Prostigmatic mites and other terrestrial arthropods of Antarctica. Antarctic Journal: 106–107.

Strenzke, K. (1955): Thalassophile und thalassobionte Collembolen. Die Tierwelt der Nord- und Ostsee 36: 52.

Strong, J. (1967): Ecology of terrestrial arthropods at Palmer Station, Antarctic Peninsula. Entomology of Antarctica 10: 357–371.

Taskaeva, A.A., Nakul, G.L. (2010): Populations of Collembola at colonies of see-gulls in Kolokolkoba Bay. In (eds. Taskaev, A.I. et al.) Actual problems of ecology and biology. Syktyvkar: 120–122.

Tejedo, P., Justel, A., Benayas, J., Rico, E., Convey, P., Quesada, A. (2009): Soil trampling in an Antarctic Specially Protected Area: tools to assess levels of human impact. Antarctic Science 21: 229–236.

Terauds, A., Chown, S.L., Bergstrom, D.M. (2011): Spatial scale and species identity influence the indigenous-alien diversity relationship in springtails. Ecology 92: 1436–1447.

Theron, P.D. (1976): new species of the genus *Terpnacarus* Grandjean (Acari: Terpnacaridae) with notes on the biology of one species. Journal of the entomological Society of South Africa 39: 131–141.

Thibaud, J.-M., Schulz, H.-J., da Gama M.M. (2004): Synopses on Palaearctic Collembola. vol. 4: Hypogastruridae. Abhandlungen und Berichte des Naturkundemuseums Görlitz 75: 1–287.

Thor, G. (1996): Mites collected in Heimefrontjella and Vestfjella (Dronning Maud Land, Antarctica). Fauna Norvegica Serie B 43: 69–73.

Tilbrook, P.J. (1967a): Arthropod ecology in the maritime Antarctic. Antarctic Research Series 10: 331–356.

Tilbrook, P.J. (1967b): Terrestrial invertebrate fauna of maritime Antarctic. Philosophical transactions of the Royal Society of London Series B-Biological Sciences 252: 261–278.

Timm, R.W. (1971): Antarctic soil and freshwater nematodes from McMurdo Sound Region. Proc. Helminthol. Soc. 38: 42–52.

Tin, T., Fleming, Z.L., Hughes, K.A., Ainley, D.G., Convey, P., Moreno, C.A., Pfeiffer, S., Scott, J., Snape, I. (2009): Impacts of local human activities on the Antarctic environment. Antarctic Science 21: 3–33.

Torricelli, G., Carapelli, A., Convey, P., Nardi, F., Boore, J.L., Frati, F. (2010): High divergence across the whole mitochondrial genome in the "pan-Antarctic" springtail *Friesea grisea*: evidence for cryptic species? Gene. 449: 30–40.

Trägårdh, I. (1908): The Acari of the Swedish South Polar Expedition. Wissenschaftliche Ergebnisse der Schwedischen Südpolar-Expedition 1901-1903 5: 1–34.

Trathan, P.F., Forcada, J., Atkinson, R., Downie, R.H., Shears, J.R. (2008): Population assessments of gentoo penguins (*Pygoscelis papua*) breeding at an important Antarctic tourist site, Goudier Island, Port Lockroy, Palmer Archipelago, Antarctica. Biological Conservation 141: 3019–3028.

Treonis, A.M., Wall, D.H., Virginia, R.A. (2000): The use of anhydrobiosis by soil nematodes in the Antarctic Dry Valleys. Functional Ecol. 14: 460–467.

Usher, M.B., Booth, R.G. (1984): Arthropod communities in a maritime Antarctic moss-turf habitat: three-dimensional distribution of mites and Collembola. Journal of Animal Ecology 53. 427–441.

Usher, M.B., Booth, R.G. (1986): Arthropod communities in a maritime Antarctic moss-turf habitat - multiple scales of pattern in the mites and Collembola. Journal of Animal Ecology 55: 155–170.

Usher, M.B., Edwards, M. (1984): The terrestrial arthopods of the grassward of Lynch Island, a specially protected area in Antarctica. Oecologia 63: 143–144.

Usher, M.B., Edwards, M. (1986a): A biometrical study of the family Tydeidae (Acari, rostigmata) in the maritime Antarctic, with descriptions of three new taxa. Journal of Zoology, London 209: 355–383.

Usher, M.B., Edwards, M. (1986b): The selection of conservation areas in Antarctica: an example using the athropod fauna of Antarctic islands. Environmental conservation. 13: 115–122.

Usher, M.B., Edwards, M. (1986c): Two new species of tydeid mite (Acarina, Prostigmata) from the maritime Antarctic. British Antarctic Survey Bulletin 73: 1–7.

VDLUFA (1991): Methodenbuch Band 1: Die Untersuchung von Böden. 4. Aufl., VDLUFA-Verlag, Speyer.

Victoria, F. de Carvalho, Pereira, A.B., Pinheiro da Costa, D. (2009): Composition and distribution of moss formations in the ice-free areas adjoining the Arctowski region, Admiralty Bay, King George Island, Antarctica. Iheringia, Sér. Bot. 64 (1): 81-91.

Vogel, M., Nicholai, V. (1983): Invertebrates collected at the old whaling station, Grytviken, South Georgia. Polar Record 135: 607–609.

Wahlgren, E. (1906): Antarktische und subantarktische Collembolen gesammelt von der schwedischen Südpolarexpedition. Wissenschaftliche Ergebnisse der Schwedischen Südpolar–Expedition, 1901–1903 (Stockholm, 1908), Zoologie 1(5): 1–22.

Wall, D.H., Adams, B.J., Barrett, J.E., Hopkins, D.W. (2006): A synthesis of soil biodiversity and ecosystem functioning in Victoria Land, Antarctica. Soil Biology & Biochemistry 38: 3001–3002.

Wallwork J.A. (1962): A redescription of *Notaspis antarctica* Michael, 1903 (Acari: Oribatei). Pacific Insects 4: 869–880.

Wallwork, J.A. (1965): The Cryptostigmata (Acari) of Antarctica with special reference to the Antarctic Peninsula and South Shetland Islands. Pacific Insects 7: 453–468.

Wallwork, J.A. (1972): Distribution patterns and population dynamics of the micro-arthropods of a desert soil in Southern California. J. Anim, Ecol. 41: 291–310.

Wallwork, J.A. (1973): Zoogeography of some terrestrial micro-arthropoda in Antarctica. Biological Reviews of the Cambridge Philosophical Society 48: 233–259.

Walter, D.E. (2001): Endemism and cryptogenesis in "segmented mites": A review of Australian Alicorhagiidae, Terpnacaridae, Oehserchestidae and Grandjeanicidae (Acari: Sarcoptiformes). Australian Journal of Entomology 40: 207–218.

Weiner, W. M. (1980): *Friesea woyciechowskii* sp. n. Collembola, Neanuridae from King George Island, South Shetland Islands, Antarctica. Polskie Pismo Entomologiczne 50: 491–495.

Wharton, D.A. (2002): Survival strategies. In: Lee, D.L. (ed.) The biology of nematodes. Taylor & Francis, London: pp. 389–411.

Wharton, D. A. (2003): The environmental physiology of Antarctic terrestrial nematodes: a review. Journal of Comparative Physiology B: Biochemical, Systemic, and Environmental Physiology 173: 621–628.

Wharton, D.A., Downes, M.F., Goodall, G., Marshall, C.J. (2005): Freezing and cryoprotective dehydration in an Antarctic nematode (*Panagrolaimus davidi*) visualized using a freeze substitution technique. Cryobiology 50: 21–28.

Whinam, J., Chilcott, N., Bergstrom, D.M. (2005): Subantarctic hitchhikers: expeditioners as ectors for the introduction of alien organisms. Biol. Conserv. 121: 207–219.

Willem, V. (1901): Les Collemboles recueillis par l'Expedition Antarctique Belge. Annales de la Société Entomologique de Belgique, 45: 260–262.

Wise, K.A.J. (1964): New records of Collembola and Acarina in Antarctica. Pacific Insects 6: 522–523.

Wise, K.A.J. (1967): Collembola (Springtails). Antarctic Research Series 10: 123–148.Wise, K.A.J. (1971): The Collembola of Antarctica. Pacific Insects 25: 57–74.

Wise, K.A.J., Fearon, C.E., Wilkes, O.R. (1964): Entomological Investigations in Antarctica, 1962 - 63 season. Pacific Insects 6: 541–570.

Witteven, J., Joose, E.N.G. (1988): The effects of inundation on marine littoral Collembola. Holarctic Ecology 1: 1–7.

Womersley, H., Strandtmann, R.W. (1963): On some free living prostigmatic mites of Antarctica. Pacific Insects 5: 451–472.

Yeates, G.W., Bongers, T., De Goede, R.G.M., Freckman, D.W., Georgieva, S.S. (1993): Feeding habits in soil nematode families and genera – an outline for soil ecologists. J. Nematology 25: 315–331.

Yue, Q.Y., Tamura, H. (2001): Three species of Collembola from Antarctic. Entomologia Sinica 8: 1–7.

Zar, J.H. (1999): Biostatistical Analysis, 4th edn. Prentice Hall, Upper Sattle River, 663 pp.

Appendix 1: Guidance documents for a standardized sampling.

Fig. A1-1: Instructions for soil-zoological sampling. In German, since all samplers were German. The instructions were developed together with all project partners before beginning the fieldwork and were provided to the expedition leaders and scientists undertaking the sampling as a guidance in choosing specific sampling sites as well as during the actual sampling.

Anleitung

Probenahme für die bodenbiologische Untersuchungen antarktischer Gebiete

1) Die verschiedenen Anlandungsorte werden im Sprachgebrauch des Vorhabens „Locations" genannt *(S. PDF-Datei, Abb.1)*

- GPS Koordinaten der Location aufnehmen (z.B. vom Schiffsausrüstung)
- Beim An-Land Gehen zuerst:
  - Gesamteindruck des Gebietes gewinnen
  - Die ökologischen Bedingungen des Gesamtgebietes abschätzen
    *→ **hieraus basiert die Auswahl der einzelnen Untersuchungsflächen („Areale", s. u.),***
  - Photos der Location nehmen (Gesamteindruck)
    Bitte alle Photos mit hoher Auflösung (z.B. 5 Megapixel) zwecks späteres digitales einzoomen
  - Dokumentation der Location laut Protokoll
    auch „nicht beprobbare" Locations (s.u.) mit Begründung (z.B. „witterungsbedingt", kein beprobbarer Substrat") protokollieren
- Beprobbare Substrat (Kritierien):
  - Alle feinere Substrate; = echte „Boden" bis Sand (= Korngröße bis 2 mm) inkl. Torf o.ä.
    Feinkies (Körngroße bis 6 mm) nur Ausnahmsweise bei einzelnen Proben
    z.B. wenn Substrat der Gesamtlocation nur aus Feinkies besteht: Location verwerfen
  - Mindest-Tiefe 10 cm
  - Location soll Vegetation (= Gras, Moos, Flechten, Bodenkrusten [= Algen]) aufweisen
    Nicht jede Untersuchungsfläche (Areal) muss Vegetation aufweisen, aber einige

2) Auswahl der Untersuchungsflächen („Areale", *s. PDF-Datei, Abb. 1***)**

Die Areale sollen dem Gesamtgebiet möglichst biologisch/ökologisch repräsentativ sein

- Es gibt 2 „Grundtypen" („**Treatments**") von Areale:

(A) Anthropogen beeinflusste Areale (v.a. Touristen, Stationspersonal)

Rote Quadrate in Abb. 1 der PDF-Datei

!! dies soll nicht in der Hauptwege der Touristenströme liegen (Boden = zu verdichtet)
eher Stellen ca. 5-10 m vom Hauptweg entfernt
wo Menschen ab und zu, aber immer wieder (z.T. „verbotenerweise") hingehen
„betreten aber nicht ausgetreten"

→ die Habitatsbedingungen dieser Treatments determinieren den zu beprobenden Habitattyp der Areale von Treatment-(B)

(B) Anthropogen unbeeinflusste Areale *(Blaue Quadrate in Abb. 1 der PDF-Datei)*

- Nie oder selten Menschen
- Auch keine offensichtliche Nist- oder Rastplätze von Tiere
- Soll der Habitattyp von (A) entsprechen (= Replikation!)

- Auswahl der Untersuchungsflächen (**„Areale"**)
  - Pro „Treatment" 3 „Areale"
  - Kriterien:
    - Beprobbare Substrat
    - Vegetation vorhanden

      Am besten alle, aber mind. 1 der drei Plots
      Vegetation kann flächendeckend sein, aber auch nur sporadisch

Appendix 1, Fig. A1-1: continued (page 2 of the instructions)

Vegetation von Treatment-(A) kann infolge von Tritt etwas (!) reduziert sein

Die verschiedenen Areale eines Treatment-Typs können sich voneinander unterscheiden!

Die Habitatsbedingungen (auch bzgl. Exposition, Beschattung usw.) der Areale von (A) und (B) sollen sich aber entsprechen (= Replikation!)

z. B.: Areal(A)a-Areal(B)a ; Areal(A)b-Areal(B)b; Areal (A)c-Areal(B)c (Abb. 1)

- Größe der Areale ca. 2-5 m x 2-5 m (je nach Bedingungen)
- Entfernung der Areale voneinander mindestens 10-20 m (je nach Bedingungen)

- Arealbeschreibung dokumentieren laut Protokoll
  Fotos der einzelnen Areale nehmen
  Falls möglich, mit Fähnchen o. ä. als Probestelle-Markeierung
  Skizze mit der Lage der Areale im Gesamtgebiet (z.B. in Karte, falls vorhanden) anfertigen

3) Auswahl der einzelnen Probestellen

- 3 Proben pro Areal
- Proben mind. 70-100cm voneinander entfernt (möglichste Dreieck-Konfiguration)
- Verteilung der Proben möglichst entsprechend des Vegetations-Mosaiks

z. B.:	*nur sehr sporadisch Veg:*	*1 Probe in Vegetation, 2 Proben aus blanker Substrat*
	patchy Vegetation:	*2 Probe in Vegetation, 1 Proben aus blanker Substrat*
	gute Vegetationsbedeckung:	*3 Probe in Vegetation, 0 Proben aus blanker Substrat*

- Dokumentation von jeder Probestelle laut Protokoll
  Skizze der Verteilung der Proben im Areal inkl. Kennzeichnung von Vegetation(Mosaik) und markante Geländemerkmale (z. B. Steine, Felsen, Ruine, Tierkolonien)
  Damit wir später die Habitatsbedingungen und Umgebung der Proben später nachvollziehen können
  Photos von jeder Probestelle (direkt von Oben)

- Zusätzlich können einzelnen Proben von interessanten Mikrohabitaten genommen werden
  z.B. Rand eines Schmelzsees oder –baches, abweichende Vegetation,
  Guano/Veg.-Boden (= ornithocoprophile Veg./Boden

4) Probenahme

- **Bodenproben** (inkl. sind darauf befindende Vegetation!)
  - Idealfall: mit Bodenstecher
    bis max. 5 cm Tiefe (von Bodenoberfläche gemessen, nicht von Vegetation!)
  - wahrscheinlicher Normalfall: mit Handschaufel
    definierte Durchmesser der Probe (Bodenstecker „simulieren“); Lineal bitte benutzen, Rund!
    Je Flacher der Boden (weniger als 5cm Tiefe), desto größer der Probe, max. 10 cm in Durchmesser

  Tiefe und Durchmesser der Probe dokumentieren (wenn unregelmäßig, mit Min. und Max.)
  Protokollieren ob Probe mit Bodenstecher oder Handschaufel genommen

- **Zusätzliche Vegetationsproben**
  Innerhalb Areal, zusätzliche Proben (nur Vegetation, z.B. Moos, Flechten)
  Anzahl der Proben abhängig von Vegetationsmosaik und „unterscheidbarer“ Arten
  [weiter von Volker]

Appendix 1, Fig. A1-1: continued (page 3 of the instructions)

5) Probebehandlung

- Bodenproben
  - Bodensubstrat (B) und darauf befindlichen Vegetation (V) der gleichen Probe in getrennten Tüten einpacken
  - Eindeutig unterscheidbare Probenummer mit Filzstift auf Plastiktüte markieren
  - Diese Nummer im Protokoll und Lage-Skizze dokumentieren
  - Plastiktüte mit Verschlussklipse verschließen
  - Kühl lagern (1-2°C) (darf nicht austrocknen oder erwärmen!)

- Zusätzliche Vegetationsproben
  - In Papiertüte (bzw. Kaffeefilter) einpacken
  - Eindeutige Kennzeichnung der Probe auf Tute
  - Dokumentieren der Probenummer in Protokoll und Lage-Skizze
  - Trocken und Luftdurchlässig bei Zimmertemperatur lagern (kann/soll austrocknen!)

Bei Fragen bitte sofort E-Mail schicken an:
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Volker.Otte@senckenberg.de

Fig. A1-2: Fieldwork protocol, to be filled in during the fieldwork by the expedition leaders and external scientists. In German, since all samplers were German. The standardized protocol served as a further guidance and ensured that all necessary data were similarly recorded by the different samplers.

Beprobungsprotokoll Nr._____ (von Blatt -1- übernehmen)

Treatment A (anthropogen beeinflusst) Fortsetzung:		**Core 1**	**Core 2**	**Core 3**	**Core 4**
Areal A-c beeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____
Treatment B (anthropogen unbeeinflusst):		**Core 1**	**Core 2**	**Core 3**	**Core 4**
Areal B-a unbeeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____
Areal B-b unbeeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____
Areal B-c unbeeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____

Beprobungsprotokoll Nr._____ **Name des Probennehmers: _____**

Datum:_____ **Name der Lokalität:_____**

GPS-Daten: _____

Geschätzte Größe der insgesamt beprobten bzw. beprobbaren Fläche: _____ x _____ m

Lufttemperatur: _____ °C **Bodentemperatur: _____ °C**

Auffälliges (*z. B. Seevogelkolonie, Schmelzwasserbach, …*) _____

Bitte fotografieren Sie zuerst das Beprobungsprotokoll (Name der Location und Datum), dann die Location (Gesamteindruck)
Notieren Sie die FotoNummern: _____
Wählen Sie für jedes Treatment (anthropogen beeinflusst /unbeeinflusst) je 3 Areale aus, die Sie mit jeweils 4 "Cores" beproben
Fotografieren Sie jedes Areal: 3 anthropogen beeinflusste, 3 anthropogen unbeeinflusste und notieren Sie Lage und Besonderheiten
Fotografieren Sie jede Beprobungsstelle senkrecht von oben und notieren Sie Durchmesser der Probenoberfläche und Tiefe

Bitte Notieren Sie: *Art u. Größe d. Beeinflussung Substrattyp, Vegetation (mit geschätzter Deckung in % der Fläche), Schwierigkeiten, Schnee, Besonderheiten, Entfernung zu Touristenhauptweg*

Achtung: Cores mind. 70 cm auseinander!

Veget: notieren Sie Art des Bewuchses
"Stecher - Schaufel": Entsprechendes ankreuzen!

Treatment A (anthropogen beeinflusst):		**Core 1**	**Core 2**	**Core 3**	**Core 4**
Areal A-a beeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____
Areal A-b beeinflusst	FotoNr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____	Nr._____ Durchmesser:____cm Tiefe:_____cm Stecher - Schaufel - ? Veget:_____ Fotonr._____

Appendix 2: Botanical results for those study areas in which plant species could be recorded at the species level. "A" = anthropogenically influenced sampling areas; "B" = non-influenced sampling areas. "Soil samples" = vegetation present on samples taken for the zoological data; "Area samples" = additional botanical samples.

Table A2-1: Arktowski-Station 2010

	Soil samples Aa	Soil samples Ab	Soil samples Ac	**Σ Soil samples A**	**A total**	Soil samples Ba	Soil samples Bb	Soil samples Bc	**Σ Soil samples B**	**B total**
Diversity determined via soil samples	X	X	X	**X**	**X**	X	X	X	**X**	**X**
Vascular Plants				**2**	**2**				**1**	**1**
Deschampsia antarctica	+	+	+	+	+	+	+		+	+
Colobanthus quietensis	+	+		+	+					
Carpet-building (pleurocarpous) Mosses				**1**	**1**				**1**	**1**
Sanionia	+	+		+	+	+	+	+	+	+
Cusion-building (acrocarpous) Mosses				**2**	**2**					
Syntrichia filaris		+		+	+					
Syntrichia magellanica		+		+	+					
Mosses total				**3**	**3**				**1**	**1**
Bryophytes total				**3**	**3**				**1**	**1**
Algae									**2**	**2**
Prasiola crispa								+	+	+
Green algae indet.								+	+	+
Total Species				**6**	**6**				**4**	**4**

Tabelle A2-2: Biologenbucht 2010

	Soil samples Aa	Soil samples Ab	Soil samples Ac	**Σ Soil samples A**	Areal samples Aa	Areal samples Ab	Areal samples Ac	**Σ Areal samples A**	**A total**	Soil samples Ba	Soil samples Bb	Soil samples Bc	**Σ Soil samples B**	Areal samples Ba	Areal samples Bc	**Σ Areal samples B**	**B total**
Diversity determined via soil samples	X	X	X	**X**					**X**	X	X	X	**X**				**X**
Diversity determined via area samples					X	X	X	**X**	**X**					X	X	**X**	**X**
Carpet-building (pleurocarpous) Mosses				**1**				**1**	**1**				**2**			**2**	**2**
Sanionia	+	+	+	+	+	+	+	+	+		+	+	+	+	+	+	+
Brachythecium austrosalebrosum										+			+	+		+	+
Cusion-building (acrocarpous) Mosses				**3**				**6**	**7**				**2**			**5**	**5**
Polytrichum alpinum			+	+	+		+	+	+						+	+	+
Ceratodon purpureus							+	+	+					+		+	+
Bryum pseudotriquetrum		+		+	+	+	+	+	+			+	+	+		+	+
Pohlia cruda														+		+	+
Syntrichia saxicola					+			+	+	+			+	+		+	+
Syntrichia filaris							+	+	+								
Bartramia patens			+	+					+								
Distichium capillaceum							+	+	+								
Mosses total				**4**				**7**	**8**				**4**			**7**	**7**
Liverworts								**1**	**1**				**1**			**1**	**1**
Cephaloziella varians					+		+	+	+	+		+	+	+		+	+
Bryophytes total				**4**				**8**	**9**				**5**			**8**	**8**
Crustose Lichens with green-algae Symbiont				**1**				**2**	**2**				**2**			**1**	**2**
Lepraria cacuminum										+			+				+
Psoroma tenue		+		+	+	+	+	+	+		+		+	+	+	+	+
Bacidia tuberculata					+			+	+								
Green-algae Lichens total				**1**				**2**	**2**				**2**			**1**	**2**
Lichens with blue-green algae symbiont				**1**				**1**	**1**				**1**			**1**	**1**
Leptogium puberulum		+		+	+	+	+	+	+		+		+		+	+	+
Lichens total				**2**				**3**	**3**				**3**			**2**	**3**
Total Species				**6**				**11**	**12**				**8**			**10**	**11**

Tabelle A2-3: Punta Cristian I 2010

	Soil samples Aa	Soil samples Ab	Soil samples Ac	**Σ Soil samples A**	Areal samples Aa	Areal samples Ab	Areal samples Ac	**Σ Areal samples A**	**A total**	Soil samples Ba	Soil samples Bb	Soil samples Bc	**Σ Soil samples B**	Areal samples Ba	Areal samples Bb	Areal samples Bc	**Σ Areal samples B**	**B total**
Diversity determined via soil samples	X	X	X	**X**					**X**	X	X	X	**X**					**X**
Diversity determined via area samples					X	X	X	**X**	**X**					X	X	X	**X**	**X**
Carpet-building (pleurocarpous) Mosses																	**1**	**1**
Sanionia														+		+	+	+
Cusion-building (acrocarpous) Mosses				**4**				**3**	**4**				**3**				**3**	**4**
Polytrichum alpinum	+	+	+	+	+	+	+	+	+		+		+	+		+	+	+
Andreaea gainii		+	+	+	+	+	+	+	+	+		+	+	+	+	+	+	+
Bryum pseudotriquetrum			+	+					+									
Bartramia patens		+	+	+			+	+	+		+	+	+					+
Ditrichum ditrichoideum														+			+	+
Mosses total				**4**				**3**	**4**				**3**				**3**	**5**
Liverworts				**1**				**1**	**1**				**2**					**2**
Cephaloziella varians			+	+			+	+	+	+	+		+					+
Lophozia excisa											+		+					+
Bryophytes total				**5**				**4**	**5**				**5**				**3**	**7**
Crustose Lichens with green-algae symbiont				**4**				**6**	**7**				**3**				**4**	**5**
Lepraria cacuminum							+	+	+					+			+	+
Psoroma tenue			+	+			+	+	+									
Psoroma hypnorum					+			+	+									
Cystocoleus ebeneus	+			+					+	+		+	+					+
Ochrolechia frigida		+		+	+	+	+	+	+			+	+	+	+		+	+
Bacidia tuberculata							+	+	+									
Placopsis contortuplicata		+		+	+	+	+	+	+			+	+	+	+	+	+	+
Rinodina olivaceobrunnea														+		+	+	+

Fruticose Lichens with green-algae Symbiont				**1**				**4**	**4**				**3**				**3**	**4**
Usnea antarctica		+		+	+			+	+			+	+					+
Usnea aurantiacoatra					+	+	+	+	+	+	+		+	+	+	+	+	+
Stereocaulon alpinum					+	+	+	+	+									
Sphaerophorus globosus														+	+	+	+	+
Himantormis lugubris					+			+	+	+			+	+	+		+	+
Green-algae Lichens total				**5**				**10**	**11**				**6**				**7**	**9**
Lichens with blue-green algae symbiont				**2**				**3**	**3**				**1**				**2**	**2**
Leptogium puberulum					+		+	+	+									
Massalongia carnosa	+		+	+			+	+	+					+			+	+
Pannaria caespitosa			+	+			+	+	+	+			+		+		+	+
Lichens total				**7**				**13**	**14**				**7**				**9**	**11**
Total Species				**12**				**17**	**18**				**12**				**12**	**18**

Tabelle A2-4: Punta Cristian II 2010

	Soil samples Aa	Soil samples Ab	Soil samples Ac	**Σ Soil samples A**	Areal samples Aa	Areal samples Ab	Areal samples Ac	**Σ Areal samples A**	**A total**	Soil samples Ba	Soil samples Bb	Soil samples Bc	**Σ Soil samples B**	Areal samples Ba	Areal samples Bb	Areal samples Bc	**Σ Areal samples B**	**B total**
Diversity determined via soil samples	X	X	X	**X**					**X**	X	X	X	**X**					**X**
Diversity determined via area samples					X	X	X	**X**	**X**					X	X	X	**X**	**X**
Carpet-building (pleurocarpous) Mosses				**1**				**1**	**1**				**1**				**1**	**1**
Sanionia spec.	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Cusion-building (acrocarpous) Mosses				**1**				**2**	**2**				**1**				**4**	**4**
Polytrichum alpinum			+	+			+	+	+		+		+		+		+	+
Bryum pseudotriquetrum					+		+	+	+					+			+	+
Bartramia patens														+	+		+	+
Chorisodontium aciphyllum														+			+	+
Mosses total				**2**				**3**	**3**				**2**				**5**	**5**
Bryophytes total				**2**				**3**	**3**				**2**				**5**	**5**
Crustose Lichens with green-algae symbiont				**1**				**3**	**3**				**4**				**5**	**6**
Lepraria cacuminum						+	+	+	+	+	+		+	+			+	+
Lepraria straminea														+	+	+	+	+
Psoroma tenue						+		+	+			+	+		+	+	+	+
Psoroma hypnorum	+	+	+	+	+		+	+	+	+	+		+					+
Ochrolechia frigida											+		+	+	+		+	+
Placopsis contortuplicata														+			+	+
Fruticose Lichens with green-algae Symbiont																	1	1
Usnea aurantiacoatra														+			+	+
Green-algae Lichens total				**1**				**3**	**3**				**4**				**6**	**7**
Lichens with blue-green algae symbiont								**1**	**1**				**1**				**2**	**2**
Massalongia carnosa											+	+	+	+			+	+
Collema spec.					+			+	+					+		+	+	+
Lichens total				**1**				**4**	**4**				**5**				**8**	**9**
Total Species				**3**				**7**	**7**				**7**				**13**	**14**

Tabelle A2-5: Ardley Island 2011

	Soil samples Aa	Soil samples Ab	Soil samples Ac	**Σ Soil samples A**	Soil samples Ba	Soil samples Bb	Soil samples Bc	**Σ Soil samples B**	Areal samples Bc	Areal samples B, (between Ac and Bc)	Areal samples B, (Area close to Ac)	**Σ Areal samples B.**	**Σ Ardley Island**
Diversity determined via soil samples	X	X	X	**X**	X	X	X	**X**					**X**
Diversity determined via area samples									X	X	X	**X**	**X**
Carpet-building (pleurocarpous) Mosses				**2**				**2**				**1**	**2**
Sanionia spec.	+	+	+	+	+	+	+	+	+	+		+	+
Warnstorfia sarmentosa	+	+		+	+			+					+
Cusion-building (acrocarpous) Mosses				**4**				**1**				**3**	**5**
Polytrichum alpinum	+		+	+			+	+	+	+	+	+	+
Ceratodon purpureus	+			+									+
Bartramia patens	+			+									+
Andreaea regularis		+		+						+		+	+
Pohlia nutans										+		+	+
Mosses total				**6**				**3**				**4**	**7**
Liverworts				**1**								**1**	**1**
Cephaloziella varians	+			+						+		+	+
Bryophytes total				**7**				**3**				**5**	**8**
Thallous Algae													
Fruticose Lichens with green-algae Symbiont				**2**				**3**				**3**	**4**
Himantormia lugubris			+	+			+	+		+	+	+	+
Usnea aurantiacoatra			+	+			+	+		+	+	+	+
Sphaerophorus globosus							+	+					+
Stereocaulon alpinum									+			+	+

Crustose Lichens with green-algae Symbiont				5				4				4	6
Lepraria cacuminum	+	+		+		+		+		+		+	+
Lepraria straminea		+		+									+
Psoroma tenue	+			+									+
Psoroma hypnorum		+	+	+		+	+	+		+		+	+
Ochrolechia frigida						+	+	+		+		+	+
Rinodina olivaceobrunnea		+		+		+		+		+		+	+
Green-algae Lichens total				**7**				**7**				**7**	**10**
Lichens with blue-green algae symbiont				1									1
Massalongia carnosa	+			+									+
Lichens total				**8**				**7**				**7**	**11**
Total Species				**15**				**10**				**12**	**19**

Tabelle A2- 6: Hannah Point 2011

	Soil samples Aa	Soil samples Ab	**Σ Soil samples A**	Soil samples Ba	Soil samples Bb	**Σ Soil samples B**	**Σ Hannah Point**
Diversity determined via soil samples	X	X	**X**	X	X	**X**	**X**
Vascular Plants			**1**			**1**	**1**
Deschampsia antarctica	+	+	**+**	+	+	**+**	**+**
Thallous Algae			**1**			**1**	**1**
Prasiola crispa	+	+	**+**	+	+	**+**	**+**
Total Species			**2**			**2**	**2**

Appendix 3: Lists of all taxa of the various soil animal groups recorded in the different study sites (= "localities") in the study years 2010 and 2011, including information on their average densities (in individuals per 100 cm^3 substrate) as well as total densities (in individuals per 100 cm^3 substrate) and total number of recorded taxa ("species") of the respective animal group. Localities are sorted from left to right by increasing southern latitude; darker shading reflects higher latitudes of a locality.

Table A3-1: Collembola

Locality	Arctowski Station (2010)	Biologenbucht (2010)	Punta Cristian (2010)	Punta Cristian 2 (2010)	Ardley Island (2011)	Halfmoon Island (2010)	Halfmoon Island (2011)	Hannah Point (2011)	Whalers Bay (2010)	Whalers Bay (2011)	Telefon Bay (2011)	Neko Harbour (2010)	Neko Harbour (2011)	Petermann Isl. (2010)	Devil Island (2010)	Paulet Island (2011)
Region	King George Isl.	King George Isl.	King George Isl.	King George Isl.	King George Isl.	Livingston Isl.	Livingston Isl.	Livingston Isl.	Deception Isl.	Deception Isl.	Deception Isl.	Antarctic Peninsula	Antarctic Peninsula	Antarctic Peninsula	Weddell Sea	Weddell Sea
Archisotoma brucei											0.04				0.7	
Cryptopygus antarcticus	102	53			37		0.1	1.9	48	0.1	8.5		4.0		9.8	0.5
Cryptopygus badasa	85	197	37	397	35			0.1	0.1		0.04				0.4	
Cryptopygus caeacus									1.3		0.2					
Folsomotoma octooculata	5.2	9.1	1.7	7.0	9.7	10.4										
Friesea grisea	5.1	9.1	0.3	0.9	2.9	15.6		1.4								
Friesea woyciechowskii	0.7	0.2	0	1.3												
Hypogastrura viatica						0.2	1.1		551	1.8	206	0.1	2.4		0.7	
Mesaphorura macrochaeta *									1.3							
Proisotoma minuta *									0.1							
Tullbergia mixta	3.8	89.7	4.1	24.8	26.1						0.2					
Total densities	202	358	43	431	111	26	1.2	3.4	602	1.8	215	0.1	6.4	0	11.5	0.5
Total number of species	6	6	3	5	5	3	2	3	6	2	6	1	2	0	4	1

* Species recorded in Antarctica for the first time

Table A3-2: Actinedida

Locality	Arctowski Station(2010)	Biologenbucht (2010)	Punta Cristian (2010)	Punta Cristian 2 (2010)	Ardley Island (2011)	Halfmoon Island (2010)	Halfmoon Island (2011)	Hannah Point (2011)	Whalers Bay (2010)	Whalers Bay (2011)	Telefon Bay (2011)	Neko Harbour (2010)	Neko Harbour (2011)	Petermann Isl. (2010)	Devil Island (2010)	Paulet Island (2011)
Region	King George Isl.	King George Isl.	King George Isl.	King George Isl.	King George Isl.	Livingston Isl.	Livingston Isl.	Livingston Isl.	Deception Isl.	Deception Isl.	Deception Isl.	Antarctic Peninsula	Antarctic Peninsula	Antarctic Peninsula	Weddell Sea	Weddell Sea
Alicorhagia sp. *									0.08							
Apotriophtydeus cf. *wilkesi*	0.07								0.09							
Apotriophtydeus scotia			11		0.86											
Apotriophtydeus sp. Juv.			4.2							0.04			0.04			
Bakerdania cf. *antarcticus*	4.9							0.06	0.47	0.98						
cf. Coccotydaeolus krantzii *				0.07				0.10	0.35	2.06			0.60	0.16		
Ereynetes macquariensis	13.1	0.34		2.8	2.4	0.11			0.26	0.04						
Eriophyidae Gen. sp.				0.09						0.08						
Eupodes (Protereunetes) sp. Juv.	1.5	0.14	1.17	0.15	0.61		0.06			0.93	0.11		1.10			
Eupodes exiguus	10.6	10.8	5.7	3.8	2.6			0.03		0.12			0.24			
Eupodes minutus	2.2	0.10	0.08	0.4		0.23				0.04						
Eupodes parvus ssp. *grahamensis*		0.13	0.74		3.98											
Heterostigmata juv.	0.22															
"Lorryia" sp.		0.13	0.09					0.06		0.12	0.05		0.04			
Nanorchestes berryi	14.5	23.7	7.6	15.21	17.4					0.16						
Nanorchestes cf. *antarcticus*											0.06					
Nanorchestes cf. *lalae*									0.16							
Nanorchestes nivalis	0.06	0.43	0.07	0.36					0.62							
Nanorchestes marianae		0.82						0.29								
Nanorchestes n.sp. *										0.51						
Nanorchestes sp. VII			0.07													

Nanorchestes sp. Juv.	0.32	10.4	1.00	0.46					0.26	1.06			0.04		
Meyerellidae/Iolinidae juv.										0.15			0.04		
Pretriophtydeus tilbrooki	0.79		0.11							0.11			0.04		
Pygmephoridae juv.									0.09	0.2					
Rhagidia gerlachei		0.11													
Rhagidia sp.	0.04														
Speleorchestes sp. *	0.52	0.09	0.26				0.05		0.41	0.12		0.16	0.04	0.16	
Stereotydeus villosus	0.13	0.10	0.15	1.71		0.12								0.16	
Stigmaeidae juv.											0.05				
Tarsonemidae juv.	0.22	0.51	0.16	0.58		0.21		0.05		0.04			0.08		0.05
Terpnacarus gibbosus *		0.19	0.09	0.09						0.04			0.12		
Total densities	**49**	**48**	**32**	**26**	**28**	**0.7**	**0.1**	**0.6**	**2.8**	**6.8**	**0.3**	**0.2**	**2.4**	**0.5**	**0.05**
Total number of species	**12**	**13**	**13**	**10**	**5**	**4**	**2**	**6**	**10**	**14**	**4**	**1**	**9**	**3**	**1**

* Taxa recorded in Antarctica for the first time

Table A3-3: Oribatida & Gamasina

Locality	Arctowski Station(2010)	Biologenbucht (2010)	Punta Cristian (2010)	Punta Cristian 2 (2010)	Ardley Island (2011)	Halfmoon Island (2010)	Halfmoon Island (2011)	Hannah Point (2011)	Whalers Bay (2010)	Whalers Bay (2011)	Telefon Bay (2011)	Neko Harbour (2010)	Neko Harbour (2011)	Petermann Isl. (2010)	Devil Island (2010)	Paulet Island (2011)
Region	King George Isl.	King George Isl.	King George Isl.	King George Isl.	King George Isl.	Livingston Isl.	Livingston Isl.	Livingston Isl.	Deception Isl.	Deception Isl.	Deception Isl.	Antarctic Peninsula	Antarctic Peninsula	Antarctic Peninsula	Weddell Sea	Weddell Sea
Oribatida																
Alaskozetes antarcticus	16					380	9.1									3.2
Halozetes belgicae	9.1															
Globoppia loxolineata	7.1		25													
Liochthonius cf. *mollis*			2.6													
Brachychochthonius sp.													1.3			
Nymphen	69		1.7			868	75	9.1								13.5
Total densities	**85**		**29**			**1247**	**84**	**9.1**					**1.3**			**16.7**
Total number of species	**3**		**2**			**1**	**1**	**1**					**1**			**1**
Gamasina																
Hydrogamasellus racovitzai	72	39	15	45	36	1.5		21								
cf. *Hydrogamasellus* sp. 2	4.4															
Parasitus tarsispinosus										30						
Genus sp. III											1.3					
Total densities	**76**	**39**	**15**	**45**	**36**			**21**		**30**	**1.3**					
Total number of species	**2**	**1**	**1**	**1**	**1**	**1**		**1**		**1**	**1**					

Table A3-4: Nematoda. Also shown are the feeding types of the individual species: al: algivore-omnivore, ba: bakterivore, ca: carnivore, fu: fungivore-radicivore.

Locality	Feeding Type	Arctowski Station(2010)	Biologenbucht (2010)	Punta Cristian (2010)	Punta Cristian 2 (2010)	Ardley Island (2011)	Halfmoon Island (2010)	Halfmoon Island (2011)	Hannah Point (2011)	Whalers Bay (2010)	Whalers Bay (2011)	Telefon Bay (2011)	Neko Harbour (2010)	Neko Harbour (2011)	Petermann Isl. (2010)	Devil Island (2010)	Paulet Island (2011)
Region		King George Isl.	King George Isl.	King George Isl.	King George Isl.	King George Isl.	Livingston Isl.	Livingston Isl.	Livingston Isl.	Deception Isl.	Deception Isl.	Deception Isl.	Antarctic Peninsula	Antarctic Peninsula	Antarctic Peninsula	Weddell Sea	Weddell Sea
Acrobeloides arctowskii	ba	230								14	209						
Amblydorylaimus isokaryon	al	4															
Aphelenchoides haguei	fu	207	1433	241	593	0.2			1		105						
Aphelenchoides helicosoma	fu	14	33	197		2											
Aphelenchoides sp. 1	fu	481															
Aporcelaimellus cf. *obtusicaudatus* *	al		6	107	4	48			0.2							0.3	
Ceratoplectus armatus	ba		1258			0.5											
Cervidellus cf. *vexilliger* *	ba	36															
Coomansus gerlachei	ca	121	57	1	7		555	124	113								
Diploscapter sp.	ba												0.1				
Ditylenchus parcevivens	fu	1070	212	59	16												
Dorylaimida sp. 1	al			16	16						73						
Dorylaimida sp. 2	al		38	61	2	4											
Enchodelus signyensis	al	73															
Eudorylaimus coniceps	al		374	47	106	54	74										
Eudorylaimus pseudocarteri	al	165	298	61		110			0.1								
Eumonhystera sp. 1	ba	335	327	36	4	0.1											
Eumonhystera vulgaris	ba	3259	327	7	315												
Filenchus sp. 1	fu		526	268	1	1											
Filenchus sp. 2	fu			91													
Geomonhystera villosa	ba		235	6	456	74			4				0.3				

Heterocephalobus sp.	ba												0.2				
Mesodorylaimus antarcticus	al	616	142	1									0.2				
Mesodorylaimus chipevi	al	348			3												
Mesodorylaimus sp. 1	al		940	86	22												
Mesodorylaimus sp. 2	al		18	291	4												
Panagrolaimus cf. *magnivulvatus*	ba	472						0.2	20	1166	93	0.2	5			306	0.2
Pellioditis cf. *marina* *	ba											354					
Pelodera cf. *strongyloides* *	ba								10				104	0.3		1378	
Plectus antarcticus	ba	41	74							421						0.3	
Plectus belgicae	ba	330	505	54	294	51			6	274	0.3				1		
Plectus insolens	ba					42											
Plectus sp. 1	ba		1460	24													
Plectus tolerans	ba	64	172	20	312	26			10								
Prismatolaimus sp.	ba	160	9	128													
Rhabditoidea sp. 1	ba						3	0.4									
Rhomborhabditis cf. *parateres* *	ba	2953									9						
Rhomborhabditis cf. *teres* *	ba								5	230	54		0.3				2309
Teratocephalus rugosus	ba	284	51			1		4									
Teratocephalus tilbrooki	ba	82	935	632	584	126	650		2		0.1					0.2	
Total densities		**11344**	**9429**	**2433**	**2739**	**830**	**1282**	**129**	**199**	**2109**	**626**	**411**	**112**	**0,8**	**1,5**	**1686**	**2310**
Total number of species		**22**	**23**	**22**	**17**	**15**	**4**	**4**	**11**	**5**	**8**	**2**	**7**	**1**	**1**	**5**	**2**

* Species recorded for the first time in Antarctica

Tabelle A3-5: Tardigrada

Locality	Arctowski Station(2010)	Biologenbucht (2010)	Punta Cristian (2010)	Punta Cristian 2 (2010)	Ardley Island (2011)	Halfmoon Island (2010)	Halfmoon Island (2011)	Hannah Point (2011)	Whalers Bay (2010)	Whalers Bay (2011)	Telefon Bay (2011)	Neko Harbour (2010)	Neko Harbour (2011)	Petermann Isl. (2010)	Devil Island (2010)	Paulet Island (2011)
Region	King George Isl.	King George Isl.	King George Isl.	King George Isl.	King George Isl.	Livingston Isl.	Livingston Isl.	Livingston Isl.	Deception Isl.	Deception Isl.	Deception Isl.	Antarctic Peninsula	Antarctic Peninsula	Antarctic Peninsula	Weddell Sea	Weddell Sea
Acutuncus antarcticus	149	0.5	0.7		1.2	471	1343	30	830	0.5		68		2.8	11	0.8
Calohypsibius sp.		0.6														
Dactylobiotus sp.	0.2				1.0											
Diphascon sp. (*Adropion*)	0.5	4.6	1.7	0.2	0.9				6.5							
Diphascon sp. (*Adropion and Diphascon*)		363	33	522	76											
Diphascon sp (*Diphascon*)		6.3	3.4	9.0	5.3											0.3
Echiniscus jenningsi		25	0.1	75.3												
Echiniscus meridionalis		0.0	84		2.3											
Hexapodibius sp.		17	0.6		0.2				0.6		0.2					
Hypsibius cf dujardini	14.6	308	0.2	70.9	46											0.7
Isohypsibius sp. 1	1.6					8.2			0.3			0.4			0.2	
Isohypsibius sp. 2		10														
Macrobiotus cf. *furciger*	202	3.0		1.5	38	348	3.3	2.2		0.1			0.3		0.3	
Pseudechiniscus sp.		0.5	11													
Ramajendas cf. *frigida*	1.1	0.4			0.2	1376	759	3.5								36
Simplex moult	0.3	24	0.3	1.2	0.3			0.6		0.6		0.3		0.7		
Total densities	**369.5**	**762.5**	**135.2**	**680.0**	**172.3**	**2202.7**	**2105.1**	**36.0**	**837.2**	**1.3**	**0.2**	**69.2**	**0.3**	**3.5**	**11.2**	**37.6**
Total number of species	**8**	**12**	**9**	**6**	**10**	**4**	**3**	**4**	**4**	**3**	**1**	**3**	**1**	**2**	**3**	**4**

Appendix 4: Spearmann rank correlations between general community parameters as well as densities of the identified soil-animal species and abiotic habitat factors as well as the vegetational cover of the various study sites. Only highly significant correlation coefficients ($P < 0.001$) are shown; negative correlations in red. Sampling date and latitude represent proxies for "locality". Zoological data used for the analyses: densities in individuals per 100 cm^3 substrate and species richness in number of species per sample.

Table A4-1: Collembola

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities	0.416	0.554	0.556		0.260					0.472		-0.266					
Species Richness	0.353	0.616	0.521		0.341		0.335			0.480							
Hypogastrura viatica	-0.520	-0.349	-0.360	0.532			-0.510	-0.410	-0.481	-0.317							
Cryptopygus antarcticus		0.553			0.457		0.288								0.307		
Cryptopygus caeacus				0.288			-0.275	-0.285	-0.287								
Folsomotoma octooculata	0.314	0.304	0.432	-0.359			0.260								-0.287		
Friesea grisea		0.547	0.399		0.420		0.452	0.273	0.416	0.361			-0.272				
Archisotoma brucei	-0.282					0.280									0.279		-0.257
Cryptopygus badasa	0.641	0.512	0.606	-0.405			0.421		0.261	0.542							
Friesea woyciechowskii	0.284																
Tullbergia mixta	0.378	0.432	0.515	-0.297			0.301			0.438							
2011																	
Total Densities		0.289	0.439		0.499	-0.296					-0.287						
Species Richness		0.326	0.479		0.479	-0.352					-0.288					0.293	
Hypogastrura viatica	-0.397																
Cryptopygus antarcticus			0.343		0.395	-0.318	0.324					0.280		-0.315		0.304	
Folsomotoma octooculata		0.439	0.438	0.363	0.426												
Friesea grisea		0.302	0.400		0.339	-0.373								-0.298	0.362	0.337	
Cryptopygus badasa		0.398	0.408	0.309	0.365												
Tullbergia mixta		0.397	0.389	0.317	0.410												

Table A4-2: Actinedida

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities	0.594	0.627	0.622	-0.358	0.310		0.521		0.355	0.596							
Species Richness	0.603	0.568	0.596	-0.320	0.283	-0.274	0.503		0.325	0.531							
Apotriophtydeus scotia				-0.287													
Bakerdania antarcticus		0.317			0.282												
Ereynetes macquariensis	0.440	0.341	0.311				0.318			0.322					-0.259	-0.276	
Eupodes exiguus	0.416	0.455	0.338	-0.288			0.471		0.327	0.437							
Nanorchestes berryi	0.556	0.449	0.523	-0.386			0.510			0.545							
Stereotydeus villosus	0.318																
Tarsonemidae			0.272														
2011																	
Total Densities		0.262	0.310	0.396	0.300			-0.291	-0.331								
Species Richness			0.259	0.364	0.256			-0.281	-0.317								
Coccotydaeolus krantzii								-0.307	-0.33								
Ereynetes macquariensis		0.311	0.306		0.323												
Nanorchestes berryi		0.455	0.441	0.400	0.405												

Table A4-3: Oribatida

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities					0.316		0.366	0.357	0.367								
Species Richness			0.256		0.327		0.333	0.323	0.346								
Alaskozetes antarcticus					0.263												
A. antarcticus+ Nymphen					0.330		0.32	0.326	0.341					-0.269			
2011																	
Total Densities	0.397		0.448	-0.515		-0.600	0.477	0.432	0.466	0.421							
Species Richness	0.404		0.453	-0.510		-0.606	0.472	0.423	0.459	0.420							
Alaskozetes antarcticus												0.328					
A. antarcticus+ Nymphen	0.408	0.299	0.463	-0.542		-0.621	0.482	0.448	0.477	0.428							

Table A4-4: Gamasina

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities	0.446	0.422	0.466				0.388		0.262	0.276							
Species Richness	0.438	0.401	0.457		0.272		0.408			0.266							
Hydrogamasellus racovitzai	0.414	0.382	0.450				0.371										
2011																	
Total Densities		0.304	0.363		0.317												
Species Richness		0.306	0.362		0.314												
Hydrogamasellus racovitzai		0.357	0.402		0.310												

Table A4-5: Nematoda

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities	0.380	0.794	0.679		0.588		0.566		0.431	0.592						0.309	
Species Richness	0.502	0.759	0.708	-0.554	0.430	-0.273	0.577		0.382	0.592	-0.276	-0.266				0.471	0.344
Acrobeloides arctowskii		0.271									0.292				0.296		
Aphelenchoides haguei		0.395	0.392	-0.294			0.296									0.285	0.270
Aphelenchoides helicosoma				-0.359													
Aphelenchoides sp. 1					0.256												
Aporcelaimellus cf. obtusicaudatus				-0.491										0.266			
Ceratoplectus armatus		0.353	0.398		0.330					0.279			-0.352		0.289	0.532	0.373
Coomansus gerlachei				-0.351		-0.286	0.336	0.344	0.357		0.288				-0.263		
Ditylenchus parcevivens		0.299	0.280	-0.338			0.340										
Dorylaimida sp. 1				-0.262													
Dorylaimida sp. 2				-0.349												0.320	
Eudorylaimus coniceps	0.327		0.427	-0.390			0.307			0.305	-0.289			0.352			
Eudorylaimus pseudocarteri		0.329	0.290	-0.312			0.315									0.326	
Eumonhystera sp. 1		0.511	0.460	-0.335	0.411		0.423		0.330	0.327						0.458	0.327
Eumonhystera vulgaris	0.575	0.546	0.541		0.351		0.410			0.473							
Filenchus sp. 1		0.421	0.415	-0.525	0.303		0.417		0.276	0.336	-0.316					0.640	0.529
Filenchus sp. 2				-0.347													
Geomonhystera villosa	0.438		0.428								-0.268	-0.360		0.360			
Mesodorylaimus antarcticus		0.359															
Mesodorylaimus chipevi		0.297			0.272				0.262		0.288						
Mesodorylaimus sp. 1		0.367	0.403	-0.400			0.289			0.373	-0.297	-0.298				0.491	0.349
Mesodorylaimus sp. 2				-0.452													
Panagrolaimus cf. magnivulvatus	-0.313			0.455													
Pelodera cf. strongyloides						0.289											
Plectus antarcticus																0.310	0.265

Plectus sp. 1		0.323	0.345	-0.278	0.280				0.264						0.481	0.314
Plectus belgicae	0.448	0.276	0.453	-0.301					0.276			0.293	0.286			
Plectus tolerans	0.524		0.308	-0.261								0.394	0.460	-0.380		
Prismatolaimus sp.		0.297		-0.390			0.306		0.295						0.266	
Rhomborhabditis cf. parateres		0.360	0.259		0.332											
Teratocephalus tilbrooki	0.403	0.431	0.526	-0.547	0.332		0.529		0.306	0.488	-0.319	-0.357			0.338	0.258
2011																
Total Densities		0.413	0.617		0.538	-0.354	0.461	0.450	0.484			0.392		-0.362		
Species Richness		0.559	0.677		0.656	-0.416	0.317		0.300					-0.410	0.403	0.282
Acrobeloides arctowskii	-0.263														0.276	
Aphelenchoides haguei	-0.109														0.284	
Aporcelaimellus cf. obtusicaudatus		0.474	0.478	0.370	0.426											
Coomansus gerlachei	0.370	0.420	0.392	-0.609		-0.687	0.433	0.384	0.416	0.481						
Eudorylaimus pseudocarteri		0.400	0.407	0.305	0.312											
Geomonhystera villosa		0.444	0.493	0.324	0.394											
Panagrolaimus cf. magnivulvatus															0.304	
Pellioditis cf. marina	-0.366		-0.257				-0.300							-0.339		
Pelodera cf. strongyloides														0.327		
Plectus belgicae		0.472	0.499	0.295	0.458											
Plectus insolens		0.439	0.447	0.363	0.424											
Plectus sp. 1																
Plectus tolerans		0.412	0.507		0.385	-0.291								0.298		
Rhomborhabditis cf. teres		-0.258					0.417	0.342	0.350			0.407		-0.380		
Teratocephalus rugosus											0.311					
Teratocephalus tilbrooki		0.579	0.603	0.411	0.541											

Table A4-6: Tardigrada

2010	Sampling Date	Latitude	Vegetational Cover	Soil Temperature (°C)	Soil Moisture (%)	Soil-pH	Organic Matter (%)	N_{tot} (%)	C_{org} (%)	C/N-Ratio	Rough Gravel	Medium Gravel	Fine Gravel	Rough Sand	Medium Sand	Fine Sand	Silt/Clay
Total Densities	0.289	0.400	0.585	-0.479	0.278	-0.300	0.420		0.379	0.367					-0.270		
Species Richness	0.475	0.511	0.660	-0.598	0.290		0.541		0.354	0.506	-0.260			0.274			
Ramajendas cf frigida				-0.264				0.292	0.284		0.305						
Acutuncus antarcticus								0.292									
Macrobiotus cf *furciger*					0.272		0.325	0.271	0.294								
Diphascon sp. *(Adr. + Diph.)*	0.422		0.407	-0.287						0.346	-0.302	-0.406		0.378			
Diphascon sp. *(Adropion)*		0.305	0.265	-0.289						0.197						0.424	0.331
Diphascon sp. *(Diphascon)*	0.465	0.260	0.426	-0.397			0.302			0.384	-0.344	-0.279		0.333			
Hypsibius cf. *dujardini*	0.368	0.396	0.504	-0.267			0.365			0.418	-0.285	-0.261					
Isohypsibius sp. 1								0.264	0.264								
Isohypsibius sp. 2																0.277	0.287
Echiniscus jenningsi	0.413		0.356							0.325				0.289			
Echiniscus meridionalis				-0.447		-0.266											
Pseudechiniscus sp.				-0.459												0.328	0.326
Hexapodibius sp.																0.342	
2011																	
Total Densities	0.315	0.653	0.720	-0.284	0.294	-0.588	0.524	0.486	0.527	0.494	0.289	0.366					
Species Richness	0.270	0.682	0.728		0.401	-0.562	0.514	0.463	0.502	0.449		0.370					
Ramajendas cf. *frigida*	0.304	0.318	0.382	-0.603		-0.538	0.550	0.500	0.525	0.514	0.365	0.399					
Acutuncus antarcticus		0.414	0.365	-0.517		-0.521	0.344	0.357	0.384	0.476							
Macrobiotus cf. *furciger*		0.335	0.356		0.257												
Diphascon sp. *(Adr. + Diph.)*		0.508	0.499	0.421	0.447												
Diphascon sp. *(Diphascon)*		0.559	0.557	0.446	0.498												
Hypsibius cf. *dujardini*		0.500	0.493	0.404	0.467												
Echiniscus meridionalis		0.292	0.282														

Appendix 5: Results of the non-parametric Friedman variance analyses (ANOVA) used to determine significant differences between localities and years (Tables 1 & 2) as well as a potential impact of human activities (Tables 3-8) on the abundances of the recorded species as well as on the total densities and species richness of the respective animal groups. The results of the study years 2010 and 2011 are given as analysed separately as well as together in Tables 3-8. Significant effects in red. „A>B“: significantly higher abundances in anthropogenically influenced areas; „B>A“: significantly higher abundances in the non-influenced areas.

Table A5-1: Differences in total densities and species richness of the various animal groups between the individual localities.

	2010		2011	
	Individuendichten			
Mesofauna	Xr^2	P	Xr^2	P
Collembola	112.806	0.001	54.932	0.001
Actinedida	86.317	0.001	58.461	0.001
Oribatida	51.520	0.001	94.321	0.001
Gamasina	43.433	0.001	33.739	0.001
Microfauna				
Nematoda	110.208	0.001	70.866	0.001
Tardigrada	74.395	0.001	116.712	0.001
	Artenzahl			
Mesofauna	Xr^2	P	Xr^2	P
Collembola	98.725	0.001	58.838	0.001
Actinedida	80.604	0.001	44.741	0.001
Oribatida	51.621	0.001	97.664	0.001
Gamasina	4.312	0.001	34.280	0.001
Microfauna				
Nematoda	140.621	0.001	84.804	0.001
Tardigrada	111.556	0.001	109.818	0.001

Table A5-2: Differences in total densties and species richness of the various animal groups (only groups of the mesofauna are schown) between study years.

Densities		
	Xr^2	p
Collembola	4.236	0.040
Actinedida	0.078	0.780
Oribatida	0.151	0.697
Gamasina	1.404	0.236
Species Richness		
	Xr^2	p
Collembola	0.851	0.356
Actinedida	0.007	0.931
Oribatida	1.785	0.182
Gamasina	0.070	0.792

Table A5-3: Total fauna. The animal groups of the meso- and microfauna were also combined for the analyses to assess possible additive effects.

Densities	2010 + 2011			2010			2011		
	Xr^2	P		Xr^2	P		Xr^2	P	
Tardigrada	5.781	0.016	B>A	10.948	0.001	B>A	0.063	0.802	
Nematoda	10.249	0.001	B>A	3.221	*0.073*	B>A	7.456	0.006	B>A
Microfauna	**1.197**	**0.274**		**0.067**	**0.796**		**1.708**	**0.191**	
Collembola	21.289	0.000	B>A	4.302	0.038	B>A	20.397	0.000	B>A
Oribatida	6.316	0.012	B>A	11.175	0.001	B>A	0.101	0.750	
Gamasina	0.466	0.495		0.191	0.662		3.469	*0.063*	B>A
Actinedida	0.051	0.821		0.140	0.708		0.007	0.931	
Mesofauna	**8.219**	**0.004**	**B>A**	**4.013**	**0.045**	**B>A**	**4.207**	**0.040**	**B>A**
Total Fauna	**0.503**	**0.478**		**0.017**	**0.897**		**0.767**	**0.381**	

Species Richness	2010 + 2011			2010			2011		
	Xr^2	P		Xr^2	P		Xr^2	P	
Tardigrada	0.066	0.797		0.033	0.857		0.035	0.852	
Nematoda	1.535	0.215		2.775	0.096		0.005	0.942	
Microfauna	**0.023**	**0.880**		**0.033**	**0.857**		**0.148**	**0.700**	
Collembola	1.497	0.221		0.738	0.390		7.270	0.007	B>A
Oribatida	8.794	0.003	B>A	9.339	0.002	B>A	1.033	0.310	
Gamasina	0.451	0.502		0.165	0.685		3.150	*0.076*	B>A
Actinedida	0.582	0.445		1.550	0.213		0.072	0.789	
Mesofauna	**0.360**	**0.549**		**0.539**	**0.463**		**2.585**	**0.108**	
Total Fauna	**0.475**	**0.491**		**0.800**	**0.371**		**0.005**	**0.946**	

Table A5-4: Collembola

	Total Effect	2010 + 2011		2010		2011	
		Xr^2	p	Xr^2	p	Xr^2	p
Total Densities	B>A	21.289	< 0.001	4.302	0.038 (B>A)	20.397	< 0.001 (B>A)
Species Richness		1.497	0.221	0.738	0.390	7.270	0.007 (B>A)
Species							
Cryptopygus antarcticu	B>A	6.842	0.009	0.489	0.484	9.202	0.002 (B>A)
Cryptopygus badasa		0.360	0.548	1.149	0.284	0.818	0.366
Cryptopygus caeacus		1.000	0.317	6.863	0.009 (A>B)	2.379	0.122
Folsomotoma octooculata		0.352	0.553	0.058	0.810	0.641	0.423
Friesea grisea		0.482	0.488	0.386	0.534	0.099	0.754
Friesea woyciechowskii				0.662	0.430	--	--
Hypogastrura viatica	B>A	15.525	< 0.001	8.383	0.004 (B>A)	7.708	0.005 (B>A)
Tullbergia mixta	B>A	7.109	0.008	1.919	0.166	8.306	0.004 (B>A)

Table A5-5: Actinedida

	Total Effect	2010 + 2011		2010		2011	
		Xr^2	p	Xr^2	p	Xr^2	p
Total Densities		0.051	0.821	0.140	0.708	0.007	0.931
Species Richness		0.582	0.445	1.550	0.213	0.072	0.789
Species							
Apotriophtydeus scotia		0.031	0.860	0.048	0.826	0.212	0.645
Bakerdania cf. *antarcticus*	*(A>B)*	3.479	*0.062*	0.033	0.855	7.754	0.005 (B>A)
Coccotydaeolus sp.				--	--	1.974	0.160
Ereynetes macquariensis	(B>A	4.264	0.039	1.515	0.218	4.242	0.039 (B>A)
Eupodes exiguus	A>B	12.481	< 0.001	12.387	< 0.001 (A>B)	0.972	0.374
Eupodes minutus				2.006	0.157	--	--
Eupodes parvus		0.320	0.572	4.267	0.039 (B>A)	1.902	0.168
Nanorchestes berryi	A>B	7.598	0.006	3.045	0.081	6.244	0.012 (A>B)
Nanorchestes gressitti				0.008	0.927	--	--
Nanorchestes n.sp.				--	--	3.273	*0.070 (A>B)*
Speleorchestes sp.		1.052	0.305	0.857	0.355	0.212	0.645
Stereotydeus villosus				0.309	0.579	--	--

Table A5-6: Oribatida & Gamasina

	Total Effect	2010 + 2011		2010		2011	
		Oribatida					
		Xr^2	p	Xr^2	p	Xr^2	p
Total Densities	B>A	6.316	0.012	11.175	0.001 (B>A)	0.101	0.750
Species Richness	B>A	9.794	0.003	9.339	0.002 (B>A)	1.033	0.310
Species							
Alaskozetes antarcticus		2.555	0.110	12.108	< 0.001 (B>A)	2.000	0.157
Globoppia loxolineata				1.466	0.226	--	
		Gamasina					
Total Densities	B>A	21.289	< 0.001	0.191	0.662	3.469	*0.063 (B>A)*
Species Richness		1.497	0.221	0.165	0.685	3.150	*0.076 (B>A)*
Species							
Hydrogamasellus racovitzai		1.702	0.300	0.021	0.885	3.208	*0.073 (B>A)*

Table A5-7: Nematoda. „↔": effects were contradictory in the two study years.

	Total Effect	2010 + 2011		2010		2011	
		Xr^2	p	Xr^2	p	Xr^2	p
Total Densities	B>A	10.249	0.001	3.221	0.073	7.456	0.006
Species Richness		1.535	0.215	2.775	0.096	0.096	0.942
Feeding types							
algivore-omnivore		2.230	0.135	0.107	0.744	10.090	0.001 (B>A)
bakterivore	B>A	6.933	0.008 (B>A)	4.194	4.194 (B>A)	2.796	0.094
fungivore-radicivore		0.100	0.752	0.163	0.686	1.999	0.157
Species							
Acrobeloides arctowskii	↔	0.677	0.410	12.670	< 0.001 (A>B)	9.035	0.003 (B>A)
Aphelenchoides haguei		1.564	0.211	0.374	0.541	2.196	0.138
Aphelenchoides helicosoma				2.941	0.086	--	--
Aporcelaimellus obtusicaudatus		0.271	0.602	0.000	1.000	0.246	0.620
Ceratoplectus armatus				1.020	0.312	--	--
Cervidellus vexilliger				1.000	0.317	--	--
Coomansus gerlachei		0.052	0.819	0.102	0.749	0.000	1.000
Ditylenchus parcevivens				0.000	1.000	--	--
Dorylaimida sp. 1	↔	0.702	0.402	10.404	0.001 (A>B)	7.149	0.008 (B>A)
Dorylaimida sp. 2				--	--	4.500	0.034 (A>B)
Enchodelus signyensis				2.000	0.157	--	--
Eudorylaimus coniceps	B>A	17.392	< 0.001	16.286	0.001 (B>A)	1.217	0.270
Eudorylaimus pseudocarteri	B>A	3.953	0.047	7.004	0.008 (B>A)	0.151	0.698
Eumonhystera sp. 1				0.763	0.382	--	--
Eumonhystera vulgaris				*3.804*	*0.051 (A>B)*	--	--
Filenchus sp. 1				0.031	0.859	--	--
Filenchus sp. 2				2.144	0.143	--	--
Geomonhystera villosa		2.710	0.100	5.324	0.021 (B>A)	0.019	0.890
Mesodorylaimus antarcticus				1.204	0.311	--	--
Mesodorylaimus chipevi				0.037	0.848	--	--
Mesodorylaimus sp. 1				0.072	0.789	--	--
Mesodorylaimus sp. 2				0.818	0.366	--	--
Panagrolaimus magnivulvatus	B>A	7.976	0.005	0.529	0.467	9.683	0.002 (B>A)

Pellioditis marina				--	--	0.507	0.477
Pelodera strongyloides		2.805	0.094	2.471	0.116	0.875	0.350
Plectus belgicae		2.881	0.090	*3.579*	*0.059 (B>A)*	0.044	0.833
Plectus insolens				--	--	5.595	0.018 (B>A)
Plectus tolerans	↔	0.454	0.500	5.666	0.017 (B>A)	5.471	0.019 (A>B)
Prismatolaimus sp.				2.015	0.156	--	--
Rhomborhabditis cf. *parateres*	B>A	7.031	0.008	*3.857*	0.050 (B>A)	3.273	0.070
Rhomborhabditis teres		1.146	0.284	3.938	0.047 (B>A)	0.085	0.770
Teratocephalus rugosus	↔	2.757	0.097	3.938	0.047 (B>A)	8.469	0.004 (A>B)
Teratocephalus tilbrooki		0.265	0.607	0.615	0.433	4.871	0.028 (A>B)

Table A5-8: Tardigrada

	Total Effect	2010 + 2011		2010		2011	
		Xr^2	p	Xr^2	p	Xr^2	p
Total Densities	B>A	5.781	0.016	10.948	0.001 (B>A)	0.063	0.802
Species Richness		0.066	0.797	0.033	0.857	0.035	0.852
Species							
Acutuncus antarcticus		0.405	0.525	0.829	0.362	4.545	0.033 (A>B)
Diphascon (Adropion + Diphascon) sp.	B>A	17.092	< 0.001	11.227	0.001 (B>A)	5.939	0.015 (B>A)
Diphascon (Adropion) sp.	*(A>B)*	*3.118*	*0.077*	2.463	0.117	1.000	0.317
Diphascon (Diphascon) sp.		0.460	0.498	0.023	0.879	0.986	0.321
Echiniscus jenningsi				0.182	0.670	--	--
Echiniscus meridionalis	B>A	19.442	< 0.001	15.541	< 0.001 (B>A)	4.430	0.035 (B>A)
Hexapodibius sp.	A>B	5.918	0.015	4.112	0.043 (A>B)	2.000	0.157
Hypsibius cf. *dujardini*		0.002	0.965	0.174	0.676	0.527	0.468
Isohypsibius sp. 2				7.149	0.008 (B>A)	--	--
Isohypsibius sp. 1				8.463	0.004 (A>B)	--	--
Macrobiotus cf. *furcige*	B>A	5.697	0.017	10.113	0.001 (B>A)	0.145	0.703
Ramajendas cf. *frigida*		1.943	0.163	1.595	0.207	0.761	0.383

Appendix 6: Results of the covariance analyses (ANCOVA) carried out to determine potential impacts by human activities (= "Anthropogenic") on the abundances of the recorded species as well as on the total densities and species richness of the respective animal groups after filtering out various habitat parameters. Separate for the study years 2010 and 2011. Also shown are detected influences of vegetational cover ("Vegetation") as well as an interaction between vegetational cover and human activity („Anthrop. x Vegetation"). Due to the danger of a Type II error (= false negative results, see Materials and Methods), only significant results are shown. „A>B": significantly higher abundances in anthropogenically influenced areas; „B>A": significantly higher abundances in the non-influenced areas. Significant effects of the vegetation always signify higher densities (or species richness) with increasing vegetational cover.

Table A6-1: Collembola. A significant influence of the vegetation on *Mesaphorura macrochaeta* exceptionally denotes higher densities in barren soils.

		Anthropogenic			Vegetation		Anthrop. x Vegetation		
Species	Year	F	P	Effect	F	P	F	P	Effect
Total Densities	2010	10.493	0.001	B>A	53.777	<0.001	3.322	0.012	B>A
	2011	5.724	0.018	B>A	8.137		2.981	0.022	B>A
Species Richness	2010								
	2011								
Cryptopygus antarcticus	2010	8.227	0.005	B>A			2.699	0.034	B>A
	2011				5.240	<0.001	2.639	0.037	B>A
Cryptopygus badasa	2010				4.767	0.001			
	2011	9.294	0.003	A>B	6.029	<0.001	7.355	<0.001	A>B
Cryptopygus caeacus	2010	4.457	0.037	A>B					
	2011								
Folsomotoma octooculata	2010								
	2011				7.191	<0.001			
Friesea grisea	2010	4.667	0.033	B>A					
	2011	8.493	0.004	B>A	6.692	<0.001	8.622	<0.001	B>A
Hypogastrura viatica	2010				20.728	<0.001			
	2011				7.917	<0.001	3.929	0.005	B>A
Mesaphorura macrochaeta	2010				2.932	0.023			
	2011								
Tullbergia mixta	2010				4.218	0.003			
	2011								

Tabelle A6-2: Actinedida

Species	Year	Anthropogenic F	Anthropogenic P	Anthropogenic Effect	Vegetation F	Vegetation P	Anthrop. x Vegetation F	Anthrop. x Vegetation P	Anthrop. x Vegetation Effect
Total Densities	2010				6.424	<0.001			
	2011								
Species Richness	2010	4.033	0.047	A>B	7.190	<0.001	2.965	0.022	A>B
	2011								
Apotriophtydeus scotia	2010	6.84	0.01	B>A			3.689	0.007	B>A
	2011								
Bakerdania cf. antarcticus	2010				6.230	<0.001			
	2011				4.107	0.003			
Coccotydaeolus sp.	2010								
	2011								
Ereynetes macquariensis	2010				3.417	0.011			
	2011						4.26	0.003	B>A
Eupodes exiguus	2010	5.986	<0.001	A>B			3.232	0.015	A>B
	2011								
Nanorchestes berryi	2010	9.478	0.003	A>B			4.995	<0.001	A>B
	2011				10.943	<0.001			

Tabelle A6-3: Oribatida

Species	Year	Anthropogenic F	Anthropogenic P	Anthropogenic Effect	Vegetation F	Vegetation P	Anthrop. x Vegetation F	Anthrop. x Vegetation P	Anthrop. x Vegetation Effect
Total Densities	2010	16.388	<0.001	B>A			3.810	0.006	B>A
	2011	8.433	0.004	A>B	48.634	<0.001			
Species Richness	2010	14.342	<0.001	B>A					
	2011								
Alaskozetes antarcticus	2010	21.577	<0.001	B>A	6.839	<0.001	7.924	<0.001	B>A
	2011	10.705	0.001	A>B	5.899		4.866	0.001	A>B

Tabelle A6-4: Gamasina

	Year	Anthropogenic F	Anthropogenic P	Anthropogenic Effect	Vegetation F	Vegetation P	Anthrop. x Vegetation F	Anthrop. x Vegetation P	Anthrop. x Vegetation Effect
Total Densities	2010				6.111	<0.001			
	2011	4.074	0.045	B>A					
Species Richness	2010				4.761	0.001			
	2011	4.823	0.030	B>A	2.482	0.046			

Tabelle A6-5: Nematoda

Species	Year	Anthropogenic F	Anthropogenic P	Anthropogenic Effect	Vegetation F	Vegetation P	Anthrop. x Vegetation F	Anthrop. x Vegetation P	Anthrop. x Vegetation Effect
Total Densities	2010				53.777	<0.001	5.362	<0.001	B>A
	2011				8.417	<0.001			
Species Richness	2010				5.824	<0.001			
	2011				10.226	<0.001	2.588	0.039	*
Acrobeloides arctowskii	2010	10.875	0.001	A>B	2.720	0.033			
	2011				4.076	0.004	3.601	0.008	B>A
Dorylaimida sp. 1	2010								
	2011				3.811	0.006			
Dorylaimida sp. 2	2010						6.146	<0.001	*
	2011								
Eudorylaimus coniceps	2010	7.238	0.008	B>A	2.687	0.034			
	2011						2.853	0.026	B>A
Eudorylaimus pseudocarteri	2010						5.743	<0.001	B>A
	2011				5.759	<0.001			
Eumonhystera vulgaris	2010	4.886	0.029	A>B			2.644	0.037	A>B
	2011	Not registered							
Geomonhystera villosa	2010				7.462	<0.001	3.383	0.012	A>B
	2011				9.234	<0.001			
Panagrolaimus cf. *magnivulvatus*	2010				5.678	<0.001			
	2011	5.204	0.024	B>A	4.860	0.001	4.133	0.003	B>A
Plectus belgicae	2010				5.120	0.001			
	2011								
Plectus tolerans	2010								
	2011				6.826	<0.001	6.746	<0.001	A>B
Rhomborhabditis cf. *parateres*	2010						3.185	0.015	*
	2011								
Rhomborhabdites cf. *teres*	2010	5.089	0.026	B>A	8.302	<0.001			
	2011	5.353	0.022	B>A	6.391	<0.001	4.172	0.003	B>A
Teratocephalus rugosus	2010						3.455	0.010	B>A
	2011	8.414	0.004	A>B	5.135	0.001	6.309	<0.001	A>B
Teratocephalus tillbrooki	2010								
	2011	16.235	<0.001	A>B			6.251	<0.001	A>B

* Higher densities in anthropogenically influenced or non-influenced areas, depending on the degree of vegetational cover.

Tabelle A6-6: Tardigrada

		Anthropogenic			Vegetation		Anthrop. x Vegetation		
Species	**Year**	**F**	**P**	**Effect**	**F**	**P**	**F**	**P**	**Effect**
Total Densities	2010				9.224	<0.001	3.614	0.008	
	2011	37.282	<0.001	A>B	13.938	<0.001	15.265	<0.001	A>B
Species Richness	2010				8.993	<0.001			
	2011	8.134	0.005	A>B	3.233	0.015	5.184	<0.001	A>B
Acutuncus antarcticus	2010								
	2011	66.906	<0.001	A>B	15.837	<0.001	18.316	<0.001	A>B
Diphascon (Adr. & *Diph.)*	2010						2.571	0.041	B>A
	2011				4.908	<0.001			
Echiniscus meridionalis	2010	8.599	0.004	B>A			3.5073	0.009	B>A
	2011	6.624	0.011	B>A	6.971	<0.001	4.888	0.001	B>A
Isohypsibius sp. 2	2010	6.149	0.014	B>A					
	2011								
Macrobiotus furciger	2010	8.411	0.004	B>A					
	2011				2.543	0.043	3.489	0.01	A>B